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Urban Forestry and Stormwater Management: Investigating the Benefit and Health of Urban Trees in Green Infrastructure Installations

Richard Andrew Tirpak
University of Tennessee

To the Graduate Council:

I am submitting herewith a dissertation written by Richard Andrew Tirpak entitled "Urban Forestry and Stormwater Management: Investigating the Benefit and Health of Urban Trees in Green Infrastructure Installations." I have examined the final electronic copy of this dissertation for form and content and recommend that it be accepted in partial fulfillment of the requirements for the degree of Doctor of Philosophy, with a major in Civil Engineering.

Jon Hathaway, Major Professor

We have read this dissertation and recommend its acceptance:

Jennifer A. Franklin, Qiang He, John S. Schwartz

Accepted for the Council:
Carolyn R. Hodges

Vice Provost and Dean of the Graduate School

(Original signatures are on file with official student records.)

Urban Forestry and Stormwater Management: Investigating the Benefit and Health of Urban Trees in Green Infrastructure Installations

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ABSTRACT

Bioretention is a green infrastructure practice commonly implemented to manage urban stormwater worldwide. While studies have described the many benefits trees provide to urban areas, including improved air quality, wildlife habitat creation, and heat island mitigation, knowledge of their contributions to stormwater management in bioretention is limited. There is a need to characterize tree health in bioretention and the performance benefits they provide to inform appropriate plant selection and maximize the functionality of these systems. In response, several studies were implemented to investigate the role of trees in bioretention practices.

The health of trees in existing bioretention practices was compared to urban trees in the southeastern United States. Using crown condition to measure overall tree health, health differences were linked to dissimilarities between bioretention conditions and species-specific site preferences. The environmental factors influencing tree health in bioretention were investigated using random forest models, which identified parameters relating to media composition and chemistry, along with species selection and planting location. Results indicated that tree health may be improved in bioretention if species selection is guided by media analysis and species compatibility with bioretention growing conditions is considered.

The contributions of various tree species in bioretention were investigated in a mesocosm-scale study. Differences in pollutant uptake between species were not significant, indicating the role of bioretention media in pollutant removal. Evapotranspiration from treed mesocosms was significantly higher than nonvegetated mesocosms, highlighting the role of transpiration in the systems. Results suggested that trees contribute to bioretention hydrology and that significant differences among species, which were attributed to growth rate, exist.

Two bioretention suspended pavement systems were installed and monitored over 27-months. Significant runoff volume reductions were observed at both practices. Influent suspended solids were significantly reduced at the underdrained practice, though other influent pollutant removal was not significant. Tree transpiration from the systems increased with greater water availability. Regression models indicated that transpiration was influenced by vapor pressure deficit and that stomatal regulation of water losses were occurring in water-limited conditions. Findings demonstrated the viability of suspended pavement systems in stormwater management applications and illustrated how design parameters influence transpiration within these practices.

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LIST OF ACRONYMS

CCSA – composite crown surface area
CCV – composite crown volume
CN – curve number
CWA – Clean Water Act
DBH – diameter at breast height
EIA – effective impervious area
ET – evapotranspiration
HRM – heat ratio method
IC – ion chromatography
ICP-AES – inductively coupled plasma – atomic emission spectroscopy
IWS – internal water storage
LA – load allocation
LID – low impact development
MDL – method detection limit
MS4 – municipal separate storm sewer system
MSE – mean square error
MTD – manufactured treatment device
NPDES – National Pollution Elimination System
NRCS – National Resources Conservation Service
OM – organic matter
OOB – out-of-bag
RCD – root collar diameter
RF – random forest
SCM – stormwater control measure
TIA – total impervious area
TMDL – total maximum daily load
TN – total nitrogen
TSS – total suspended solids
UN – United Nations
USDA – United States Department of Agriculture
USEPA – United States Environmental Protection Agency
USFS – United States Forest Service
WLA – waste load allocation

CHAPTER 1 : INTRODUCTION

Urban stormwater runoff has been recognized as one of the leading causes of aquatic habitat degradation in the United States and in cities throughout the world (USEPA, 2018a). As urbanization and land-use conversion intensifies, impervious surfaces that are found extensively in urban areas, such as roads, parking areas, and rooftops, alter the hydrology and quality of stormwater runoff to the detriment of receiving waterbodies (Arnold and Gibbons, 1996, Walsh et al., 2005). The development of stormwater-specific regulations that apply to a growing number of cities and municipalities across the United States has led to the widespread implementation of alternative strategies, such as green infrastructure stormwater control measures (SCMs), to manage urban stormwater runoff (USEPA, 2018b). One such SCM, the bioretention practice, has been widely implemented due to its versatile design and demonstrated success in mitigating the impacts of urban runoff (Davis et al., 2009).

Bioretention practices typically consist of an excavated area of land backfilled with a sandy engineered soil media underlain by drainage rock and topped with mulch and a variety of vegetation, including grasses, shrubs, and bushes (Hunt et al., 2012). Numerous studies have characterized the ability of bioretention practices to lessen the impacts of stormwater runoff through peak flow mitigation, volume reduction, sedimentation and filtration of suspended particulates, and adsorption of dissolved pollutants to soil particles (e.g., Davis et al., 2003, Davis et al., 2012, Li and Davis, 2009, Hatt et al., 2007, Hunt et al., 2006, Olszewski and Davis, 2013). Further studies have shown that plants contribute to the management of runoff in bioretention practices through evapotranspiration (ET) and nutrient uptake (Lucas and Greenway, 2009, Read et al., 2008). While plant species have demonstrated varying levels of contributions to bioretention performance, relatively little information for plant selection based on physiological characteristics exists (Read et al., 2008). As a result, many studies and design guidelines are limited to vegetation types selected on their ability to withstand the unique growing conditions found in bioretention practices, such as grasses, sedges, and shrubs. Only recently have a few studies investigated the role of trees in these systems (Denman et al., 2016, Scharenbroch et al., 2016).

Trees provide a number of ecosystem services to the urban environment which have been characterized by a growing body of literature. Urban trees mitigate the heat island effect via evaporative cooling and shading and improve air quality through the interception of particulate matter and adsorption of gaseous pollutants (Nowak et al., 2006, Taha et al., 1989). Trees influence hydrology as rainfall reaching the canopy is partitioned into interception, stemflow, and throughfall (Xiao and McPherson, 2016). In addition to these environmental benefits, urban trees may play an important role in the management of urban stormwater runoff in bioretention practices. However, until the potential role of trees in these systems is more fully understood, there is little incentive for designers and regulators to promote their inclusion in bioretention practices.

To address this research need, several studies were conducted at the University of Tennessee between 2015 and 2018 to characterize the role of trees in bioretention practices. The health of trees in existing bioretention practices were compared to other similar urban trees in five cities in the southeast United States using crown condition as an indicator of overall tree health. A mesocosm-scale study was designed to investigate potential differences in hydrologic and water quality performance contributions of various tree species in bioretention practices.

Two bioretention suspended pavement systems were installed to assess their performance in urban stormwater management at the field-scale. Finally, a study using sap flow sensors was conducted to examine the influence of site conditions and meteorological factors on tree-water dynamics in bioretention systems. The overall objective of this research is to characterize the role of trees in bioretention practices and identify physiological aspects and design parameters that influence their success and contribution to the management of urban stormwater runoff in these systems. The principal research question being addressed by this dissertation is:

What role do trees serve in the management of urban stormwater runoff in bioretention practices, and how can tree species be selected based on physiological aspects that optimize their contributions to the hydrologic and pollutant removal performance of bioretention practices?

This dissertation is separated into chapters focusing on individual components which contribute to the overall objective of this research. Chapter 2 provides a review of relevant background literature on stormwater management, bioretention practices, and urban trees. Chapter 3 consists of a published manuscript on the health of trees in bioretention practices and the environmental factors that influence tree health. Chapter 4 describes the mesocosm-scale study examining the performance contributions of trees in bioretention practices. The performance study of two bioretention suspended pavement systems is the subject of Chapter 5. Chapter 6 contains a published manuscript on the influence of site conditions and meteorological factors on tree-water dynamics in bioretention suspended pavement systems. Reference sections for individual chapters are provided at the end of each of these chapters. The dissertation culminates with a review of conclusions drawn from this research along with recommendations for future work, which are found in Chapter 7.

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CHAPTER 2 : LITERATURE REVIEW

This chapter provides a review of literature on urban stormwater management to provide background for the research project. The review begins with an overview of urban stormwater management, starting with the environmental impacts of urbanization. The development of regulations relevant to stormwater and the evolution of urban stormwater management practices to mitigate these impacts in the United States are then discussed. The next section provides an overview of bioretention design components and the treatment processes involved in urban stormwater management in bioretention practices. This is followed by a review of hydrologic and water quality performance of bioretention practices reported in literature, along with a discussion of the role of vegetation in bioretention practices. Research on the role of trees in the urban environment is also discussed. Finally, the chapter culminates in a review of existing knowledge gaps in the field and outlines the specific aims and objectives of this research.

2.1 Urban Stormwater Management

2.1.1 Urbanization

Urbanization refers to the increasing concentration of a nation's population dwelling in urban areas (USEPA, 2017). The 2014 revision of the United Nation's World Urbanization Prospects report found that the global population in urban areas (54%) had surpassed that of rural areas for the first time, and that population growth in urban areas will continue to rapidly increase for the next several decades, especially in developing nations (United Nations, 2014). Historically, urbanizing populations have been associated with socioeconomic advances, such as greater life expectancy, higher levels of education and literacy, lower fertility, improved living conditions, and increased geographic and social mobility (Satterthwaite et al., 2010). However, increased urban populations and land development pose significant challenges to the environment at various spatial scales. As a result of concentrated development and industrial activity, urban areas are the primary source of atmospheric greenhouse gas emissions that drive global climate change, accounting for 78% of carbon emissions worldwide (Brown, 2001). Demand for agricultural production, consumption of natural resources, land-use change, and waste generation associated with increased human populations in urban areas influence an array of biogeochemical processes at the regional scale (Grimm et al., 2008).

Locally, urban areas influence air temperatures through a phenomenon known as the urban heat island effect, where warmer surface temperatures derived from the prevalence of urban building materials impact micro-scale climate patterns, influence heat emissions, and drive energy consumption (Revi et al., 2014). Urbanization also has a dramatic impact on local streams and waterways. Extensive land-use conversion characteristic of urban areas influences the hydrology and water chemistry of aquatic ecosystems by increasing the total amount of runoff produced and the rate at which runoff is delivered to urban streams, altering streambank stability and channel morphology, increasing water temperatures, diminishing aquatic habitats, biodiversity and species populations, and introducing pollutants unique to the urban environment in concentrated loads (Leopold, 1968, Walsh et al., 2005). Sprawling urban areas increasingly intersect and impact stream networks, and the number of stream systems degraded by urban development will likely continue to grow as urbanization increases throughout the world (Meyer et al., 2005). The proportion of impervious surfaces in urban centers can serve as a metric of receiving water bodies impacts due to the linkage between key characteristics of urbanization (i.e., land-use conversion and concentrated human activity) and urban stream degradation.

2.1.2 Impervious Cover and Deviation from Natural Landscape Function

Impervious surfaces, such as roads, parking areas, and rooftops, have long been connected to changes to the urban hydrologic cycle and the associated impacts to local streams and receiving water bodies (Leopold, 1968, Hollis, 1975, Klein, 1979). These surfaces, which constitute a high percentage of land in urban areas, influence the natural hydrology of the landscape by preventing the infiltration of water into underlying soils (Arnold and Gibbons, 1996). Further alterations to natural surfaces that occur during urban development, including soil compaction and vegetation clearing, add to the reduced capacity of the urban landscape to absorb water, restore groundwater supply, and mitigate runoff conveyance to streams (Booth and Jackson, 1997). As such, a greater portion of rainfall is converted to runoff that, coupled with the efficiency with which runoff is transported via structures such as gutters and pipe networks, contributes to a variety of degraded stream conditions collectively referred to as the urban stream syndrome (Walsh et al., 2005). Symptoms of the urban stream syndrome related to impervious surfaces include decreased base flows, increased occurrences of flood events, decreased time to peak flow rates, increased flow duration, diminished water quality, straightening and deepening of stream channels, and reduced aquatic habitat complexity and biodiversity (Walsh et al., 2005).

Research has indicated that distinguishing the degree of connectivity between the impervious area and the stream network can provide a more robust quantification of the impact of urbanization on stream networks. Though the total impervious area (TIA) in a city was historically used as an indicator of development, it provides a limited comparison of the impact of land development between watersheds. For example, the level of TIA in a watershed may include impervious surfaces which are routed to pervious areas and thus will not contribute any excess runoff to a stream channel or sewer network or create any significant hydrologic changes to the system (Booth and Jackson, 1997). Instead of TIA, Booth and Jackson suggest that the impact of land development on a watershed should be characterized by the effective impervious area (EIA), defined as "impervious surfaces with direct hydraulic connection to the downstream drainage (or stream) network" (Booth and Jackson, 1997). As opposed to TIA, essentially all stormwater generated from EIA will reach streams and other receiving water bodies (Brabec et al., 2002). Thus, streams in watersheds with higher levels of EIA will be subjected to larger impacts from stormwater runoff and lead to the subsequent degradation of urban stream conditions. The widespread alteration and degradation of waterways in the United States has led to the development of regulations aimed to mitigate the environmental impacts of urbanization and impervious land cover.

2.1.3 Evolution of US Stormwater Regulations

Though urban stormwater runoff has been recognized as a contributing factor to the impairment of streams for some time, lessening the impacts of stormwater in the United States has posed a series of challenges to regulatory agencies. The difficulty in regulating stormwater impacts "arises from three basic attributes of what is commonly termed 'stormwater':

1. [Stormwater] is produced from literally everywhere in a developed landscape;
2. [Stormwater] production and delivery are episodic, and these fluctuations are difficult to attenuate; and
3. [Stormwater] accumulates and transports much of the collective waste of the urban environment." (NRC, 2009).

As a result, stormwater-specific federal regulations in the United States have only been developed and implemented over the last thirty years and continue to evolve as the scientific understanding of urban stormwater impacts improve. The following sections review the predominant environmental laws that provide for the development and implementation of stormwater regulations in the United States.

2.1.3.1 Clean Water Act

Created as a 1972 amendment to the Federal Water Pollution Control Act of 1948 after mounting public pressure to remediate the nation's waters, the Clean Water Act (CWA) established the foundation for pollutant discharge regulation for waters in the United States (USEPA, 2017b). The goal of the CWA was to restore and maintain the integrity of US waters by eliminating non-permitted pollutant discharges to waterways and establishing water quality standards to sustain aquatic life and provide for human recreation (USEPA, 2017b). Under the CWA, the United States Environmental Protection Agency (USEPA) was granted the authority to implement pollution control programs and determine water quality standards for surface water contaminants (USEPA, 2017b). Though originally focused solely on point-source discharges, the CWA recognized the importance of non-point source pollutant discharges, enabling the creation of future stormwater regulations through successive legislation.

2.1.3.2 National Pollutant Discharge Elimination System (NPDES)

The original passage of the CWA in 1972 created the National Pollutant Discharge Elimination System (NPDES) which implemented a permitting system to restrict point-source industrial and municipal pollutant sources into US waterways (USEPA, 2017c). The 1987 amendments to the CWA directed the USEPA to regulate large industrial and municipal stormwater discharges by establishing a permit system and implementing discharge standards, which was completed in a series of two installments. In 1990, the USEPA Phase 1 Stormwater

Permit Rules were implemented, which applied to various industrial sectors, construction sites greater than five acres, and municipal separate storm sewer systems (MS4s) serving a population greater than 100,000 residents (NRC, 2009). Phase 2 of the NPDES Stormwater Permit Rules were released in 1999 and applied to MS4s serving smaller populations (as defined by the US census) and construction projects greater than one acre in size (USEPA, 2005). Phase 2 permits also require MS4s to develop, implement, and enforce stormwater management programs designed to reduce pollutant discharge (USEPA, 2005). The NPDES Phase 2 Rule outlines six "minimum control measures" that must be included in MS4 stormwater management programs, including: public education and outreach, public participation and involvement, illicit discharge detection and elimination, construction site runoff control, post-construction runoff control, and pollution prevention and good housekeeping (USEPA, 2005).

2.1.3.3 Clean Water Act Section 303(d) and Total Maximum Daily Loads (TMDL)

Section 303(d) of the CWA requires states to evaluate available water quality data and identify waterways that do not meet water quality standards established in the CWA, otherwise known as impaired or threatened waters (USEPA, 2017d). Section 303(d) further requires states to report lists (termed "303(d) lists") of impaired waters along with the pollutant(s) causing the impairment and report to the USEPA every two years (USEPA, 2017d). Once the state has identified impaired waterways and the pollutants of concern, it must develop total maximum daily loads (TMDLs) that are prioritized based on the level and severity of pollution and the types of applications for which the waters are used (USEPA, 2017d). TMDLs for each waterway/pollutant combination on a state's 303(d) are determined by calculating the maximum amount of a pollutant that could be discharged into the water (including background concentrations and a factor of safety) while still meeting water quality standards (USEPA, 2017e).

After TMDLs are established, pollutant allocations are assigned to point sources in the form of waste load allocations (WLA) and non-point sources in the form of load allocations (LA), which are submitted to the USEPA for approval (USEPA, 2017e). Once TMDLs are approved for a waterway, it is taken off the 303(d) list; however, TMDLs continue to be tracked until the water is fully restored (USEPA, 2017d). The process of tracking 303(d) lists and establishing TMDLs provides a critical framework for states to identify impaired waters, create and implement plans to rehabilitate waters by regulating pollution sources, and track the progress of restoration to ensure all waters meet the necessary water quality standards.

2.1.4 Evolution of Stormwater Management Approaches

Many strategies and approaches to stormwater management have been implemented in cities over the course of several millennia. Structures designed to control flooding, convey wastewater, and store rainwater for future use were found in cities of the Mesopotamian Empire, dating back to the second millennium BC (NRC, 2009). Covered drains and sewer networks helped prevent flooding and transport used water away from urban areas in ancient Rome (Sedlak, 2014). As technology and the scientific understanding of hydrology advanced over time, the approaches used in modern-day stormwater management began to take form. However, the original objective of urban stormwater management established in ancient societies, to convey water away from the urban landscape as quickly as possible to prevent flooding, remained well into the 20th century (Fletcher et al., 2015).

Modern urban drainage systems, featuring networks of pipes and catch basins that quickly conveyed stormwater to surrounding waterbodies, appeared in American cities following World War II (NRC, 2009). Shortly thereafter, in response to the channel degradation and erosion issues that resulted from the efficiency with which these systems transported runoff, stream channels were commonly widened and lined with concrete to maintain flood attenuation capacity (NRC, 2009). With the rise of the environmental movement in the 1960s and 1970s, societal concerns for the impact on aquatic habitats, which were widely impacted by changes imparted to downstream waters by straightened, lined conveyance channels, began a rapid evolution in the approach to managing stormwater (Figure 2.1) (Fletcher et al., 2015).

The first modification to urban stormwater management for the purposes of mitigating environmental impacts came in the form of on-site detention basins in the 1970s (NRC, 2009). On-site detention basins were implemented to reduce peak flow rates of runoff produced from selected storm sizes before it exited the boundary of the developed area (NRC, 2009). However, stormwater detention basins still contributed to flooding-related issues on the watershed scale. While site-scale peak flows were controlled through the slow release of detained stormwater, individual outflows were independent of other practices in the watershed, and the unchanged volume of stormwater and cumulative outflow rate still led to further stream degradation (McCuen, 1979, Ferguson, 1991). Stormwater management approaches began to address the volume control limitations of detention basins through the advent of low impact development (LID) in the 1990s (Fletcher et al., 2015). LID techniques focus on the goal of maintaining a site's pre-development, natural hydrology through the promotion of infiltration/groundwater recharge and mitigating the volume and frequency of stormwater discharges (NRC, 2009). LID

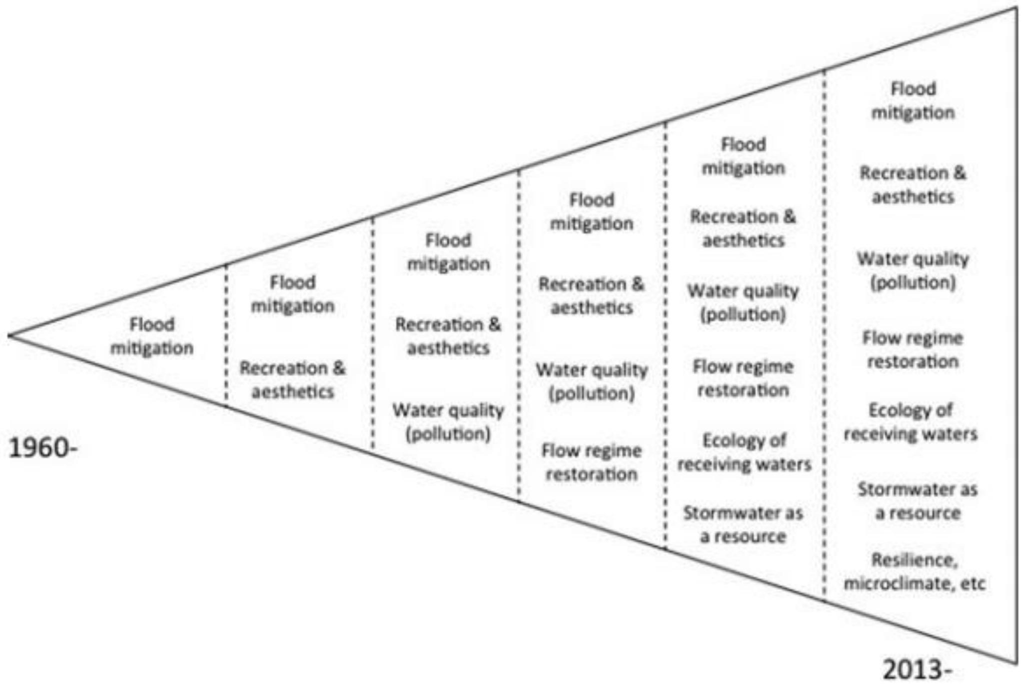

Figure 2.1: Evolution of modern-day approaches to urban stormwater management (Fletcher et al., 2015, adapted from Whelans et al., 1994).

principles target stormwater runoff management near the source of production through distributed, small-scale stormwater devices such as bioretention systems, disconnected rooftop drainage, permeable pavement, and grassed swales (NRC, 2009, Fletcher et al., 2015).

In conjunction with the evolution of design strategies to mitigate the hydrologic impacts of urban runoff, concerns about the chemical composition of stormwater being delivered to receiving waterbodies led to the incorporation of water quality improvement as a major objective of urban stormwater management (NRC, 2009). Contaminants of concern commonly associated with urban stormwater runoff are total suspended solids (TSS), bacteria (fecal coliform), nutrients (nitrogen and phosphorous compounds), hydrocarbons, heavy metals (such as copper, lead, and zinc), and other toxic substances (USEPA, 1983). Sources of these pollutants in urban areas include construction activities (TSS), fertilizers and pesticides (nutrients), animal waste (nutrients, fecal coliform), atmospheric deposition (TSS, nutrients, heavy metals), industrial activity (heavy metals, hydrocarbons), and automobiles (TSS, hydrocarbons, heavy metals) (Brown, 2011). Today's stormwater management approaches combine both hydrologic and water quality goals, such as mitigating peak flow rates and/or reducing stormwater volumes and achieving specified pollutant removal levels (which can vary on a state-by-state basis), to restore pre-development hydrologic regimes and the ecological function of receiving waterbodies (NRC, 2009, Fletcher et al., 2015). Innovative approaches referred to as stormwater control measures (SCMs) are increasingly being implemented by cities and municipalities in the United States to meet these stormwater management objectives.

SCMs are techniques, measures, or structures designed and implemented to manage the quantity and improve the quality of urban stormwater runoff in a particular set of conditions

(NRC, 2009). SCMs can take the form of both structural and non-structural forms to manage urban stormwater (Fletcher et al., 2015). Structural SCMs encompass built or engineered physical infrastructure, such as bioretention practices, green rooftops, permeable pavement, etc., while non-structural SCMs refer to preventative efforts to lessen the impact of urban runoff, such as public outreach and education programs, consideration for stormwater runoff in land-planning and site design activities, and downspout disconnection programs (Fletcher et al., 2015, NRC, 2009). Though they may differ in size, scope, and applicability, structural SCMs are designed with the intent of achieving pre-development hydrologic conditions and utilizing natural treatment mechanisms to mitigate the impact of urban stormwater runoff quantity and quality on receiving waterbodies.

2.2 Role and Function of Bioretention Practices

2.2.1 Bioretention Design Components

Bioretention practices, also referred to as bioinfiltration practices, biofilters, or rain gardens, are one of the most widely used structural SCMs throughout the United States and many other parts of the world (Davis et al. 2009). Though designs vary, bioretention practices typically consist of an engineered sandy soil media underlain by a layer of drainage rock and topped with turf grass or mulch and various forms of vegetation, including shrubs, trees, and grasses (Figure 2.2).

Stormwater runoff produced from a contributing drainage area is routed into the bioretention practice, where it temporarily fills the surface storage zone (referred to as the bowl) before infiltrating into the system. Several processes contribute to the hydrologic and water quality management of stormwater inside of the practice, including sedimentation, filtration, soil

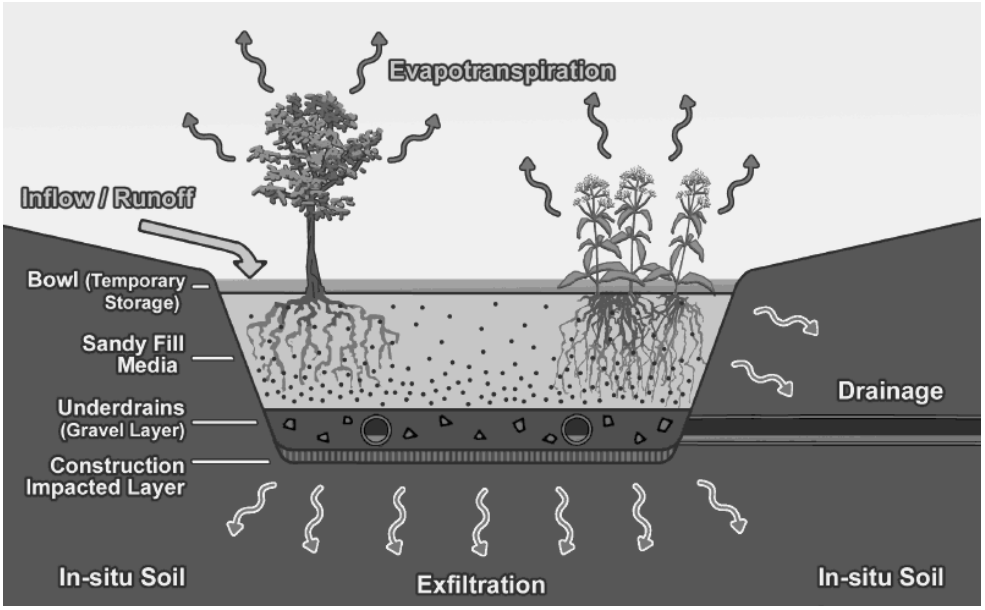

Figure 2.2: Cross-section of a typical bioretention cell with drainage network connection (Hunt et al., 2012, image created by S. Kennedy, NC State University)

adsorption, and biological assimilation (Davis et al., 2009). Once the runoff has infiltrated through the bioretention profile, it can exit the system through evapotranspiration (ET), exfiltration into underlying subsoils, or, when present, be collected in perforated pipe networks and transported into downstream drainage infrastructure. In order to maintain the functionality of these systems, bioretention practices must be regularly maintained to sustain media permeability through activities such as debris clearing and removal, and mulch and plant replacement when necessary (Hunt and Lord, 2006).

Many of the design components of a bioretention practice in Figure 2.2 can be modified to meet specific hydrologic and water quality management goals, regulatory objectives, and/or site constraints (Hunt et al., 2012). Depending on design criteria and site conditions, bioretention practices may or may not contain perforated underdrain systems to transport treated stormwater. Another popular design parameter includes a constantly maintained saturated zone of media at the bottom of the practice, referred to as the internal water storage (IWS) layer, created by elevating the level of the underdrain. The intent of the IWS layer is to create anaerobic conditions in a portion of the bioretention media to promote nitrogen removal via denitrification, though previous studies have reported mixed levels of performance (e.g., Dietz and Clausen, 2006, Hunt et al., 2006, Passeport et al., 2009, Brown and Hunt, 2011a). Incorporating an IWS layer in design has also been shown to provide additional runoff storage for volume reduction within the bioretention practice (Li et al., 2009, Winston et al., 2016). Though design variations have evolved since their inception, the design versatility and established performance have led to the widespread recognition of bioretention practices as an effective SCM for urban stormwater management.

2.2.2 Stormwater Treatment Performance of Bioretention Practices

2.2.2.1 Hydrology

Over the last two decades, a growing body of research has demonstrated the benefits of urban stormwater management using bioretention practices. From a hydrologic perspective, bioretention practices have been shown to effectively reduce runoff volumes. In many instances, runoff from small events is completely captured by bioretention practices and produces no outflow to downstream waters because of soil porosity, exfiltration, and temporary bowl storage (Davis et al., 2012). Through complete abstraction of runoff from these events, bioretention practices can have significant impacts on the total amount of runoff produced from urban areas. Further, when outflow from a bioretention practice does occur, peak flow rates and runoff volumes are often significantly reduced. Hydrologic results and key findings from select bioretention performance studies are summarized in Table 2.1. These results, along with numerous other studies, collectively illustrate the impact that bioretention practices can have on the hydrology of a developed urban area.

2.2.2.2 Water Quality

As with the hydrologic impacts, several studies have characterized the water quality benefits of bioretention practices. Water quality results from select bioretention performance studies are summarized in Table 2.2. Bioretention practices are highly effective at reducing influent TSS concentrations through sedimentation and filtration processes that take place as runoff percolates through the bioretention media (Hunt et al., 2012). Adsorption at complexation locations, particularly at iron and aluminum oxide deposits, in bioretention media provide for high removal rates of heavy metals, such as copper, lead, and zinc, due to the relatively low

Table 2.1: Hydrologic results from previous bioretention literature. Values are presented as percent reduction (%).

Study	Peak Flow Reduction	Volume Reduction	Key Hydrologic Findings
Brown and Hunt, 2011a	-	75-100	Volume removal via ET and exfiltration increased with deeper IWS zone depths (75% removal when IWS thickness was 0.73m compared to 87% removal at 1.03m); highest volume reduction observed when sites were installed on sandier underlying soils
Davis, 2008	49-58	-	18% of rainfall events did not produce outflows; inclusion of an IWS layer (Cell B) improved the probabilities of meeting peak flow and runoff reduction goals
DeBusk and Wynn, 2011	99	97	82% of storms did not produce outflows; volume reduction may have been attributed to larger media depth (0.6-1.2m) and lateral seepage into surrounding soils
Hunt et al., 2006	-	>50	Significantly higher outflow volumes occurred in winter months compared to summer; outflow reductions highly influenced pollutant removal performance
Li et al., 2009	-	20-50	40% of events produced zero outflow; deeper media profiles (>0.9m) and larger practice surface areas enhanced performance; ET was a substantial hydrologic pathway (19% of water losses)
Olszewski and Davis, 2013	83	79	Curve number (CN) comparisons used to determine how bioretention flows compared to predevelopment hydrology; analysis suggested that larger surface areas were needed to meet hydrology of a wooded space
Winston et al., 2016	24-96	36-59	Highest peak flow reductions observed when adequate bowl storage was available to prevent overflow; volume reduction was greatest in the practice with the highest IWS zone thickness and largest drawdown rate

metal content of urban runoff and the neutral pH range of the media (Hunt et al., 2012, Wang et al., 2017). Particle-bound phosphorous (P) in runoff is primarily removed through filtration, while dissolved P species are removed through chemical sorption to amorphous iron and aluminum present in bioretention media (Hunt et al., 2012). P removal by bioretention practices has varied over time and has been associated with evolutions in bioretention media composition. Clark and Pitt (2009) found that P (and N) could be leached from high organic content bioretention media, while Hunt et al. (2006) associated poor P removal with high P-index media. Studies have shown improved P removal through low organic matter (OM), low P-index bioretention media (e.g., Bratieres et al., 2008, Hatt et al., 2009a, Hunt et al., 2008, Passeport et al., 2009).

The majority of nitrogen (N) removal in bioretention practices occurs through bacterially mediated nitrification-denitrification pathways (Hunt et al., 2012). Ammonium (NH_4^+) in runoff is first converted to nitrite (NO_2^-) and then to highly mobile nitrate (NO_3^-) under aerobic conditions (Hunt et al., 2012). The relatively high infiltration rate of bioretention media creates largely aerobic soil conditions, thus ammonium is readily converted to nitrogen oxides (NO_x), typically resulting in low effluent concentrations (Table 2.2) (Brown, 2011). Denitrification, however, requires anaerobic conditions to be established to remove N from the bioretention practice through the conversion of NO_x to nitrogen gas (N_2) (Hunt et al., 2012). Because exposure to anaerobic conditions is so critical and is inconsistently established in bioretention media, NO_x removal often fluctuates between practices. Deeper media depths can provide opportunities for the creation of anaerobic zones and enhanced N removal (Hunt et al., 2012). IWS layers and low OM content media specifications (though enough to act as an electron donor) have been used to create anaerobic conditions in the bottom layers of bioretention media that favor denitrification, though N removal remains varied. Vegetation in bioretention, which will be discussed in the following section, has also been shown to improve N removal through root uptake and assimilation (Bratieres et al., 2008, Lucas and Greenway, 2008).

2.2.3 Role of Vegetation in Bioretention Practices

Several studies have demonstrated the benefits that many types of vegetation have contributed to urban stormwater management in bioretention practices. A hydrologic benefit attributed to vegetation in bioretention practices is runoff volume reduction as water is removed from bioretention media via plant transpiration (ET). Brown and Hunt (2011b) reported that ET accounted for 3% of total runoff volume reduction from seven bioretention practices planted with shrubs, trees, and perennials. Sharkey and Hunt (2005) found that 50% of influent runoff was exported from a bioretention cell in Louisburg, NC through ET. Conversely, Hess et al. (2017) used weighing lysimeters in rain gardens planted with switch grass, perennials, and deciduous shrubs in three media types with varied drainage configurations to conclude that ET accounted for between 43% and 70% of the water budgets. The differences in ET were attributed to the inclusion of an IWS layer, which enhanced removal in columns planted with a sandy soil media compared to similar configurations where an unrestricted drainage configuration was used (Hess et al., 2017). Though they have been shown to provide significant volume reduction via ET, plant selection and local site conditions appear influence the degree to which plants impact bioretention hydrology. The discussion of ET in bioretention is continued in dissertation Chapters 4-6.

Table 2.2: Water quality data from previous bioretention literature. Values are presented as percent removal (%).

Study	TSS	TN	NH_4^+-N	NO_x-N	TP	Cu	Pb	Zn
Bratieres et al., 2008 (approx. average of trials)	>95	70	-	-	85	-	-	-
Brown and Hunt, 2011a	58	58	74	58	-10	-	-	-
Chapman and Horner, 2010 (110th Cascade Method A)	87	63	-	-	67	80	86	80
DeBusk and Wynn, 2011	>99	>99	-	-	>99	-	-	-
Davis et al., 2003	-	-	-	-	-	43	70	64
Davis, 2007	47	-	-	-	76	57	83	62
Geheniau et al., 2015	75	-	-	-	-65	-14	54	48
Hatt et al., 2007	>80	-	-	-		>90	>90	>90
Hatt et al., 2009a (McDowall site)	93	37	96	-17	86	98	98	99
Hatt et al., 2009a (Monash site)	76	-7	64	-13	-398	67	80	84
Hatt et al., 2009b (Clayton site)	76	-7	64	-13	-398	67	80	84
Hunt et al., 2006 (C1 site)	-	40	-	11	65	-	-	-
Hunt et al., 2006 (G2 site)	-170	40	-	82	-240	99	81	98
Hunt et al., 2008	60	32	73	-	31	54	31	77
Li and Davis, 2009 (CP site)	94	23	-	17	34	50	70	89
Li and Davis, 2009 (SS site)	92	64	-	-	-	57	33	93
Muha et al., 2016 (Bioretention 1)	98	77	-	-	86	-	-	-
Muha et al., 2016 (Bioretention 2)	97	74	-	-	84	-	-	-
Passeport et al., 2009 (North site)	-	56	-	78	53	-	-	-
Passeport et al., 2009 (South site)	-	57	-	88	68	-	-	-
Randall and Bradford, 2013	-	53	-	-	76	-	-	-
Smolek et al., 2018	92	33	13	-97	66	-10	-	66

In addition to volume reduction through ET, plant root systems can also help to maintain soil structure and hydraulic conductivity of bioretention media. Lewis et al. (2008) found that hydraulic conductivity of a bioretention practice planted with native rushes and sedges recovered after initial soil compaction and settling post-construction due to plant root growth and the formation of macropores in the bioretention media. Similarly, Hatt et al. (2009a) saw increases in infiltration rate coincide with vigorous plant growth in a bioretention practice densely planted with rushes and sedges and indicated the important role of vegetation and root systems in maintaining bioretention media porosity and soil structure to preserve system function over time. Le Coustumer et al. (2012) found that hydraulic conductivity values in bioretention columns planted with *M. ericifolia*, a tall, thick-rooted shrub, increased during a 72-week trial, while other more thinly-rooted plants, which can form dense mats that act as choking layers within the media profile, did not maintain media permeability levels.

Vegetation has also been shown to improve the water quality performance of bioretention practices by contributing to the removal of heavy metals found in urban stormwater runoff. Muthanna et al. (2007) studied water quality performance in a pilot-scale bioretention box and found that between 2% and 7% of heavy metal removal could be attributed to the shrubs and flowering species planted in the systems via assimilation into roots and leaves. However, in a column study of the effectiveness of 20 Australian monocot and dicot species, Read et al. (2008) did not find any significant differences (on average) between vegetated and nonvegetated columns in the removal of TSS and heavy metals, though effluent concentrations were generally low for all configurations. Feng et al. (2012) found that plants in bioretention columns planted with shrubs, grasses, sedges, and perennials significantly influenced the removal of iron (Fe), chromium (Cr), and aluminum (Al), however Cu, Pb, and Zn levels were unaffected by the presence of vegetation. Contributions of metal removal varied between species for Fe, Cr, and Al, and, in-line with the nutrient performance results attributed to the species in other studies, it was recommended that future practices utilize *Carex appressa*, a tall native grass, due to its demonstrated removal level for the metals studied (Feng et al., 2012).

Finally, plants in bioretention practices have been shown to contribute to N and P removal from urban runoff. Lucas and Greenway (2008) reported higher levels of TN, NO_x, and TP removal in bioretention mesocosms with different soil compositions compared to nonvegetated systems. The shrubs and grasses used in the study were estimated to contribute 6%, 47%, and 35% of overall TP, NO_x, and TN removal, respectively (Lucas and Greenway, 2008). Plants significantly lowered effluent N and P concentrations on average compared to soil-only controls in column trials conducted by Read et al. (2008), though as high as 20x variation in removal between species was observed. Plant size and root mass had significant influence on treatment contributions, and species were not consistent in reducing all pollutants in stormwater, leading to the recommendation that mixed planting palates be utilized in future bioretention practices (Read et al., 2008). Of the 20-species examined, several grasses (including *Carex appressa*), shrubs, and rushes were identified as preferential selections after standardizing for root mass (Read et al., 2008).

Bratieres et al. (2008) found that the presence of vegetation had a large effect on NO_x and TN removal in bioretention columns. Again, *Carex appressa* provided the best N and P removal compared to other species due to the formation of an extensive root system containing microscopic root hairs that increased the volume of soil available to the plant (Bratieres et al., 2008). Zhang et al. (2011) found that the presence of rushes, shrubs, and sedges significantly influenced TN and NH_4^+-N reductions compared to nonvegetated bioretention columns, but not NO_x-N or TP removal, and that vegetated column performance was improved in the presence of a submerged zone (IWS). Payne et al. (2014) used isotopic nitrogen to determine how NO_3^- in stormwater runoff was partitioned inside of bioretention mesocosms and found that plant assimilation served a key uptake pathway of influent nitrate. Nitrate assimilation varied among plant species, though, on average, several grass species were able to assimilate 89%-99% of incoming nitrate present in synthetic stormwater runoff (Payne et al., 2014).

Relatively few studies have examined the performance of trees in bioretention practices compared to other types of vegetation. Denman et al. (2016) studied the nutrient removal contribution of four street trees (including both native evergreen and exotic deciduous species) in 240mm diameter bioretention columns with varying hydraulic conductivities. The presence of trees significantly improved soluble P removal, though no differences were observed between vegetated and nonvegetated columns in low hydraulic conductivity trials (Denman et al., 2016). NO_x removal was also improved by the presence of trees, though significant differences between species and soil types were mixed and varied seasonally (Denman et al., 2016). The authors concluded that while trees reduced NO_x and P relative to unplanted controls, species selection did not influence nutrient removal performance (Denman et al., 2016). Turk et al. (2017) investigated nutrient removal performance of pairs of native and common cultivar versions of various plants, including two tree species (sweet bay magnolia, *Magnolia virginiana* L., and river birch, *Betula nigra* L.) in a number of bioretention practices in North Carolina. Analysis of leaf, root, and shoot material approximately one year after planting found that N and P uptake from the two *B. nigra* varieties ranged between 172.5-209.0g N and 15-15.5g P, far higher than any other species used in the study (Turk et al., 2017). Similarly, N and P removal for *M. virginiana* varieties ranged between 10.3-21.2g N and 0.8-1.2g P (Turk et al., 2017). By factoring in canopy cover, planting costs, and nutrient uptake, *B. nigra* varieties provided the greatest nutrient removal benefit per areal cost, accounting for 1668-2020 g $\$^{-1}$ m^{-2} N and 145-150 g $\$^{-1}$ m^{-2} P (Turk et al., 2017).

One bioretention design configuration gaining popularity in recent years is manufactured treatment devices (MTDs), also called tree box filters or tree pits, which generally consist of a small concrete form filled with bioretention media installed alongside roads or parking areas and planted with a single tree (Geronimo et al., 2014, Smolek et al., 2018). A benefit of these devices is their small footprint, which make them a preferred option over traditional bioretention practices in highly-developed urban areas when land availability is limited (Smolek et al., 2018). Geronimo et al. (2014) reported high levels of TSS (80-98%) and heavy metal removal (up to 70%) from an MTD near a parking lot, though specific contributions resulting from the presence of the tree in the system were not reported. Similarly, Smolek et al. (2018) evaluated the performance of a MTD planted with a crepe myrtle (*Lagerstroemia* spp.) alongside a parking area in Fayetteville, North Carolina. Overall, the authors found that the MTD was effective in treating stormwater from a small impervious watershed (water quality results presented in Table 2.2), though no treatment performance contributions specific to the tree were investigated, indicating the need for further research in this area (Smolek et al., 2018).

2.3 Role and Trees in the Urban Environment

The environmental, social, and economic benefits of trees in urban areas has been widely studied and recognized for several decades. The following presents findings from a limited amount of the vast body of literature that has demonstrated the various benefits of trees in urban areas. Ecosystem services provided by urban trees include: mitigation of the urban heat island effect, reduced gaseous pollutant emissions and improved air quality, improved human health, increased habitat for wildlife, and hydrologic benefits such as rainfall interception, stemflow, and transpiration (Figure 2.3).

Urban forests mitigate the effects of the urban heat island effect through shading and evaporative cooling, and as a result reduce energy consumption and associated cooling costs in nearby buildings (Livesley et al., 2016). Kurn et al. (1994) found that near-surface air temperatures under vegetated canopies were 3-4°C lower than background air temperatures in Los Angeles, CA. Taha et al. (1989) reported that average daytime temperatures inside of an urban tree canopy in Davis, CA were 1.5°C cooler than surrounding areas. In a more recent study, climatic models created by Ballinas and Barradas (2016) predicted that mature urban forests could reduce the temperature in Mexico City by 1°C, though species selection influenced the planting density and number of trees required to meet this target. Akbari (2002) calculated that urban trees can reduce cooling costs and energy use by 25% through shielding direct solar radiation from buildings, reducing the radiation of heat from nearby surfaces toward buildings, and lowering nearby air temperatures through shading and evaporative cooling.

Another benefit of trees in urban areas is improved air quality and reduced pollutant levels, which can have positive effects on human health. Through the interception of particulate

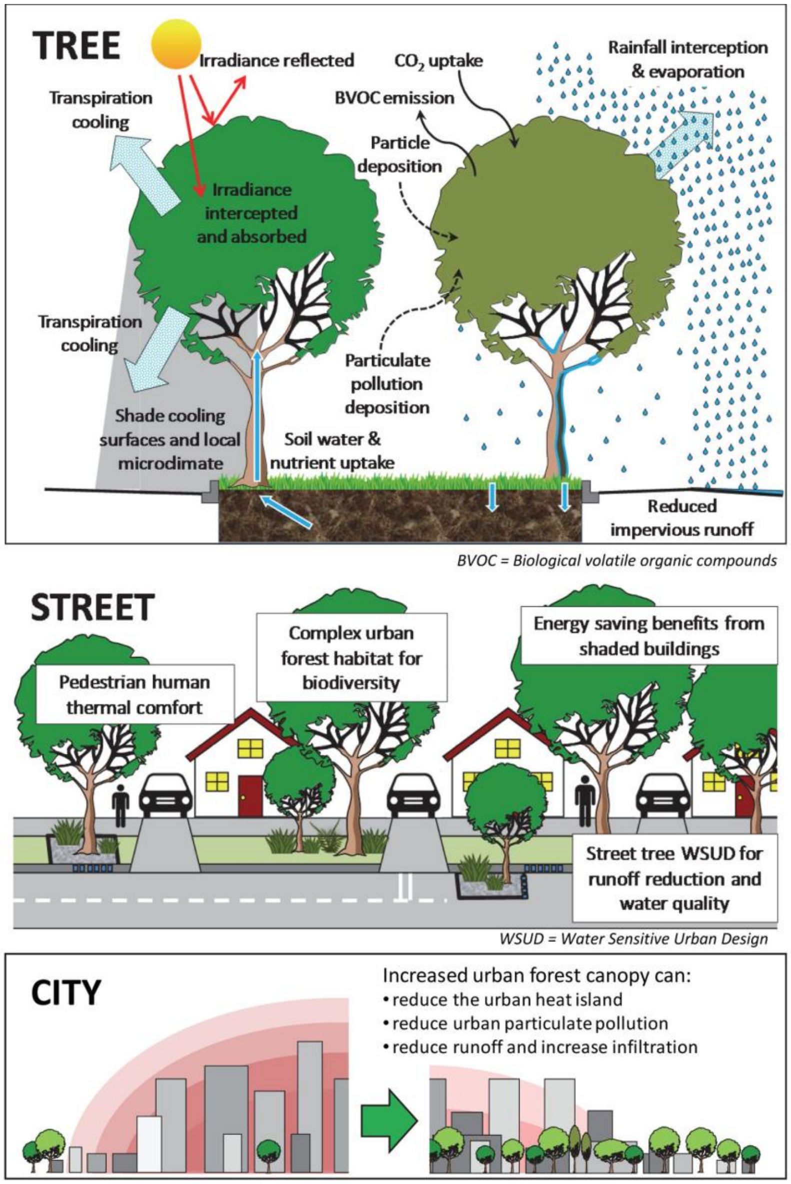

Figure 2.3: Ecosystem services provided by urban trees at the individual tree, street, and city scale (adapted from Livesley et al., 2016).

matter on vegetated surfaces and adsorption of gaseous air pollutants (e.g., CO_2, CO, NO_2, SO_2, and O_3), models by Nowak et al. (2006) estimated that urban trees in the United States removed 711,000 metric tons of air pollutants. In a modelling assessment of the benefits of street trees in California, McPherson et al. (2016) reported that trees in the state removed 567,748 metric tons of CO_2, equivalent to eliminating emissions from 120,000 automobiles, along with 1,358 tons of ozone and 772 tons of particulate matter per year. Stands of trees alongside streets also provide habitat for wildlife, attenuate vehicle noise, and protect pedestrians from motorists (Mullaney et al., 2015). Though rural areas were included in the analysis, Nowak et al. (2014) reported that the human health benefits related to the avoidance of respiratory-related illnesses and conditions derived from trees in the contiguous United States in 2010 were valued at $6.8 billion, most of which was realized in urban areas.

Trees also provide hydrologic benefits to urban areas through processes such as interception, stemflow, throughfall, improved infiltration, and transpiration, all of which vary between species and are dependent on local site conditions. Tree interception consists of the portion of rainfall landing on a tree that is retained in the canopy or bark and eventually evaporates from the tree surface, and thus never contributing to surface runoff (Berland et al., 2017). Tree interception is influenced by a number of species-specific and meteorological factors, including: leaf area, leaf size and storage capacity, leaf and bark texture, branch architecture, temperature, relative humidity, wind speed, rainfall intensity and duration, and net radiation (Xiao and McPherson, 2016). Generally, a tree will intercept a greater portion of rainfall from longer duration, lower intensity rainfall events during conditions which favor evaporation (i.e., higher temperatures and wind speeds) than shorter, more intense rain events

(Xiao and McPherson, 2016). Results from studies of interception assembled by Kuehler et al. (2017) reported that between 6% and 82% of rainfall was retained as interception for various tree species in urban or open-grown conditions.

Once the crown and bark storage become fully saturated, rainfall begins to reach the ground surface and contribute to surface runoff through stemflow and throughfall. Stemflow refers to the portion of rainfall that is initially intercepted by the canopy but then flows down the stems, branches, and trunk to the ground surface, while throughfall is the portion of rain that falls through the canopy and travels directly to the ground (Xiao and McPherson, 2011). Both stemflow and throughfall influence urban hydrology by reducing and delaying the delivery of peak runoff to the ground surface (Xiao and McPherson, 2000, Asadian and Weiler, 2009). Additionally, these processes offer protection for soils beneath canopies from potential erosive forces from direct rainfall, maintaining soil structure and infiltration capacity to further reduce surface runoff (Asadian and Weiler, 2009).

In addition to providing aboveground runoff storage via interception, trees influence hydrology by promoting infiltration along root channels that have penetrated through compacted urban soils. Day et al. (2000) studied the performance of flowering dogwood (*Cornus florida* L.) and silver maple (*Acer saccharinum* L.) in various combinations of soil strength and water tension and found that roots of the bottomland species, *A. saccharinum* L., grew moderately well in compacted soils with high water content, while *C. florida* L. was unsuccessful. Bartens et al. (2008) found that roots of both black oak (*Quercus velutina* Lam.) and red maple (*Acer rubrum* L.) penetrated clay loam soils at two compaction levels and increased average infiltration rates by 153%. Improving infiltration rates of heavily compacted urban soils further decreases the amount of stormwater runoff that results in overland surface flow.

Finally, trees influence urban hydrology by converting stormwater runoff that has infiltrated into soils to water vapor that is released to the atmosphere via transpiration. In addition to reducing the overall volume of stormwater, the process of transpiration regenerates soil water holding capacity, promoting further runoff retention and storage (Kuehler et al., 2016). Like interception, stemflow, and throughfall, the hydrologic contributions of tree transpiration vary depending on several factors, such as: incoming solar radiation, temperature, humidity, size and intensity of rainfall, soil moisture conditions, species selection, planting density, leaf area, etc. Pataki et al. (2011) used constant-heat sap flow sensors to study tree transpiration and their water use requirements in the dry climate of Los Angeles, CA. After scaling to the plot scale, large differences in species transpiration were found, ranging from averages of 3.2 kg $tree^{-1}$ d^{-1} for Canary Island pine (*Pinus canariensis*) to 176.9 kg $tree^{-1}$ d^{-1} for London plane tree (*Platanus hybrid*) (Pataki et al., 2011). Scharenbroch et al. (2016) monitored the impact that a variety of tree species grown in bioswales had on the hydrology of a parking lot in Illinois. Using monthly average stomatal conductance measurements to model transpiration, Scharenbroch et al. (2016) found that tree transpiration varied between species and accounted for 46% to 72% of the total water balance of the system and recommended that species with large mature size and greater total leaf area will likely contribute more toward system hydrology. With these studies for context, transpiration is a highly varied yet critical mechanism through which trees influence urban hydrology and is further addressed in dissertation Chapters 4-6.

2.4 Knowledge Gaps and Research Contributions

The presence of vegetation, species selection, plant size, root development, and soil water conditions within bioretention have been shown to serve an important role in the hydrologic and water quality performance of these practices. However, while some studies draw comparisons between species contributions, many studies of bioretention vegetation are limited to grasses, shrubs, sedges, and other hardy plant forms that were selected based on their ability to withstand the dramatic soil moisture conditions found in bioretention media (i.e., prolonged soil dryness with periods of soil inundation during rain events). This leaves a need to explore the viability of other plant types in bioretention practices as well as the development of a physiology-based approach to plant selection to improve bioretention performance. Further, while a small number of studies have investigated the performance contributions of trees in bioretention, results comparing treatment performance benefits between species and linkages to physiological aspects that may account for these differences are limited or nonexistent.

Though several studies have characterized the environmental, social, hydrologic, and economic benefits of trees in urban areas, the role of urban trees in bioretention practices has not yet been thoroughly explored. As a long-lived plant form with extensive above-ground and below-ground biomass, trees have the potential to improve on the hydrologic and water quality aspects of bioretention performance previously reported for other forms of vegetation. Characterizing the benefits of trees in bioretention practices and identifying physiological aspects that are connected to tree performance will provide urban foresters and stormwater engineers with critical information that will allow them to select the most appropriate plants to maximize bioretention functionality and stormwater treatment. Promoting and optimizing the use of trees in bioretention practices will also incorporate the various ecosystem services attributed to urban trees, increasing the overall environmental benefits of the bioretention practice.

To more fully understand the role of trees in bioretention practices, these knowledge gaps were addressed with several targeted research efforts. The following chapters aim to accomplish the following:

1. Evaluate the health of trees currently planted in bioretention practices relative to other urban trees, and investigate specific bioretention design parameters that influence tree health;
2. Characterize the hydrologic and water quality benefits of various tree species in bioretention practices and investigate physiological aspects that may be linked to their performance contributions;
3. Assess the performance of tree-specific suspended pavement devices designed to function as subsurface alternatives to bioretention, examine the influence of bioretention design parameters on tree function, and quantify the contribution of urban trees to bioretention performance at the field scale; and
4. Develop design guidelines and recommendations to help urban foresters, stormwater engineers, and regulatory agencies understand how to best integrate trees into bioretention practices and quantify their contributions to urban stormwater management.

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CHAPTER 3 : THE HEALTH OF TREES IN BIORETENTION: A SURVEY AND ANALYSIS OF INFLUENTIAL VARIABLES

A version of this chapter was originally published by R. Andrew Tirpak, Jon M. Hathaway, Jennifer A. Franklin, and Anahita Khojandi:

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R. Andrew Tirpak prepared the following chapter as part of his Ph.D. dissertation under the supervision of co-authors Jon M. Hathaway, Jennifer A. Franklin, and Anahita Khojandi, who provided editorial assistance and text contributions to the manuscript.

3.1 Abstract

Bioretention is a commonly used stormwater control measure that, through biogeochemical processes, can improve water quality and reduce runoff volume generated from impervious surfaces. Vegetation has been shown to improve bioretention treatment performance and lifespan, yet guidance for plant selection in bioretention systems remains relatively general, particularly for trees. While numerous benefits of urban trees are understood, including heat island mitigation, air quality improvement, and the like, knowledge of their potential contributions to stormwater management as a component of bioretention is minimal. Critical to tree function in these systems is the trees' ability to maintain health in the unique substrate and hydrologic regime found in the bioretention environment. This study investigated tree health in bioretention systems in the southeastern United States using three-dimensional composite indicators of crown volume and surface area. Five tree species were found to be in a less-healthy state when planted in bioretention practices compared with similar urban trees, whereas only bald cypress (*Taxodium distichum*) exhibited greater health in bioretention. Differences in tree health were attributed to a lack of alignment between typical bioretention conditions and species-specific growing preferences. Regression models were created using random forest methods to identify bioretention parameters that impact tree health. Parameters relating to bioretention

media composition, media chemistry, and tree species selection and planting location (upslope, midslope, or bottom of the bioretention system) were found to have the most influence on tree health. Results from this study suggest that tree health in bioretention may be improved if species selection is based on bioretention media analysis and consideration of species compatibility with the growing conditions found in bioretention.

3.2 Introduction

As a result of land use conversion associated with development and pollutants generated through concentrated anthropogenic activity, urban stormwater runoff and related water management issues, such as combined sewer overflow events, are a significant cause of water quality impairment and degradation in aquatic environments (USEPA, 1999). Cities and municipalities across the United States and beyond are increasingly turning toward green infrastructure stormwater control measures (SCMs), especially bioretention, as an urban stormwater management strategy (Davis et al., 2009). Bioretention systems, also known as biofilters or rain gardens, are excavated landscape depressions that are backfilled with a sandy soil media and usually topped with mulch and various types of vegetation. The intent of these systems is to mimic the hydrologic and water treatment processes that occur in the natural environment (Davis et al., 2009, Hunt et al., 2012). Various design modifications have been made to enhance bioretention performance, including selecting media compositions that mitigate runoff flows and improve infiltration, limiting phosphorous (P) content in media for increased P removal, and installing internal water storage (IWS) zones to create anaerobic conditions and improve nitrogen removal via denitrification (Davis et al., 2009, Davis et al., 2012, Hunt et al., 2006, Brown and Hunt, 2011). Although previous research has demonstrated the critical influence of plants on bioretention hydrologic and water quality performance (Barrett et al., 2013, Feng et al., 2012, Hatt et al., 2009, Lucas and Greenway, 2008, Read et al., 2008, Bratieres et al., 2008), few design modifications have been proposed that specifically promote plant health and function in bioretention systems. Instead, plants are often selected on their ability to survive the extreme soil moisture fluctuations and nutrient-deficient environments found in bioretention systems. Much of the existing research is limited to grasses and hardy shrubs/sedges that can tolerate such conditions, and few studies have examined the specific role of trees in bioretention systems (Denman et al., 2016, Hart, 2017).

Conversely, extensive research has demonstrated that urban trees provide a number of critical ecosystem services to metropolitan areas worldwide. Hirokawa (2011) and Young (2011) describe a number of studies that have linked urban trees to improved air quality, mitigation of the urban heat island effect, benefits to human health, increased property value and wildlife habitat, and energy conservation through shading and evaporative cooling. Urban tree canopies influence stormwater runoff through rainfall interception, while their root systems improve infiltration, limit soil erosion, and regulate soil nutrient cycles involved in stormwater pollutant removal (Xiao and McPherson, 2011, Bartens et al., 2008, Day et al., 2010, Kuehler et al., 2017). As with other types of vegetation, little research has been conducted to identify the potential contributions urban trees may bring to stormwater management in bioretention systems, or how the unique environmental conditions found in bioretention systems influence tree health and function (Denman et al., 2016, McPherson et al., 2011, Scharenbroch et al., 2016). Thus, the extent to which current bioretention designs, species selection, and planting/maintenance practices contribute to tree health and promote various environmental benefits associated with healthy trees is unknown.

Although limited consideration is given to plant health in bioretention cells, the health status of vegetation is essential for optimal system function. Due to their size and rooting volume, trees may have great impacts on bioretention performance, and knowledge of tree function in these systems is needed to promote sustainable urban systems. This study reports on field assessments of tree health in 38 bioretention areas in Tennessee and North Carolina conducted during the summer of 2015. Tree health was quantified using measurement and classification of crown condition (Schomaker et al., 2007). Because tree crowns play a vital role in photosynthate generation and net primary production, their dimensions and fullness can be used as indicators of general tree health (Zarnoch et al., 2004).

Several environmental factors common to bioretention systems may contribute to poor tree health and function. Trees that are not tolerant of quick-draining, low-nutrient soils may not have access to enough nutrients and water to sustain healthy, vigorous growth. Conversely, the pollutants unique to the water chemistry of urban stormwater runoff, such as heavy metals, hydrocarbons, and other chemicals, may be present at toxic levels in bioretention media and have corresponding adverse effects on tree health and condition. The purpose of this study is to assess the overall health of trees planted in bioretention areas compared with other analogous urban trees and to investigate which basic design parameters and species-specific growing preferences are most influential to tree health in bioretention systems. Findings from this study can be used in design specifications and guidelines to maximize tree health in bioretention systems and promote the inclusion of trees in future bioretention installations.

3.3 Materials and Methods

3.3.1 Site Descriptions and Experimental Setup

The cities of Nashville and Chattanooga in Tennessee, and Raleigh, Cary, and Durham in North Carolina were selected for this study to encompass a range of climatic and geographical characteristics representative of many urban areas in the inland southeastern United States. Stormwater officials from each location were consulted to identify bioretention systems containing trees and to obtain site access permission from landowners prior to commencing field activities. In total, 38 bioretention systems containing trees were identified for this study. These systems varied in several design components (surface area, media composition, available ponding depth, drainage configuration, proximity to other infrastructure, tree size, tree count per practice, tree species, and the like), and design plans were obtained and consulted when possible to confirm design versus as-built system conditions. Figure 3.1 illustrates the metropolitan areas included in the study, while Table 3.1 provides a summary of the bioretention systems and their locations. It should be noted that other parameters that may influence tree health in bioretention, such as catchment size and imperviousness, were not available for this study.

3.3.2 Tree Health Using Composite Crown Indicators

Using methodology adopted by the United States Forest Service (USFS), two independent observers standing one tree length away from the stem rated eight absolute crown condition indicators (vigor class, uncompacted live crown ratio, crown light exposure, crown position, crown density, crown dieback, foliar transparency, and crown diameter) for two perpendicular cross sections of each tree crown. Although each indicator reflects a different aspect of the crown, larger, more densely vegetated crowns generally correspond to vigorous,

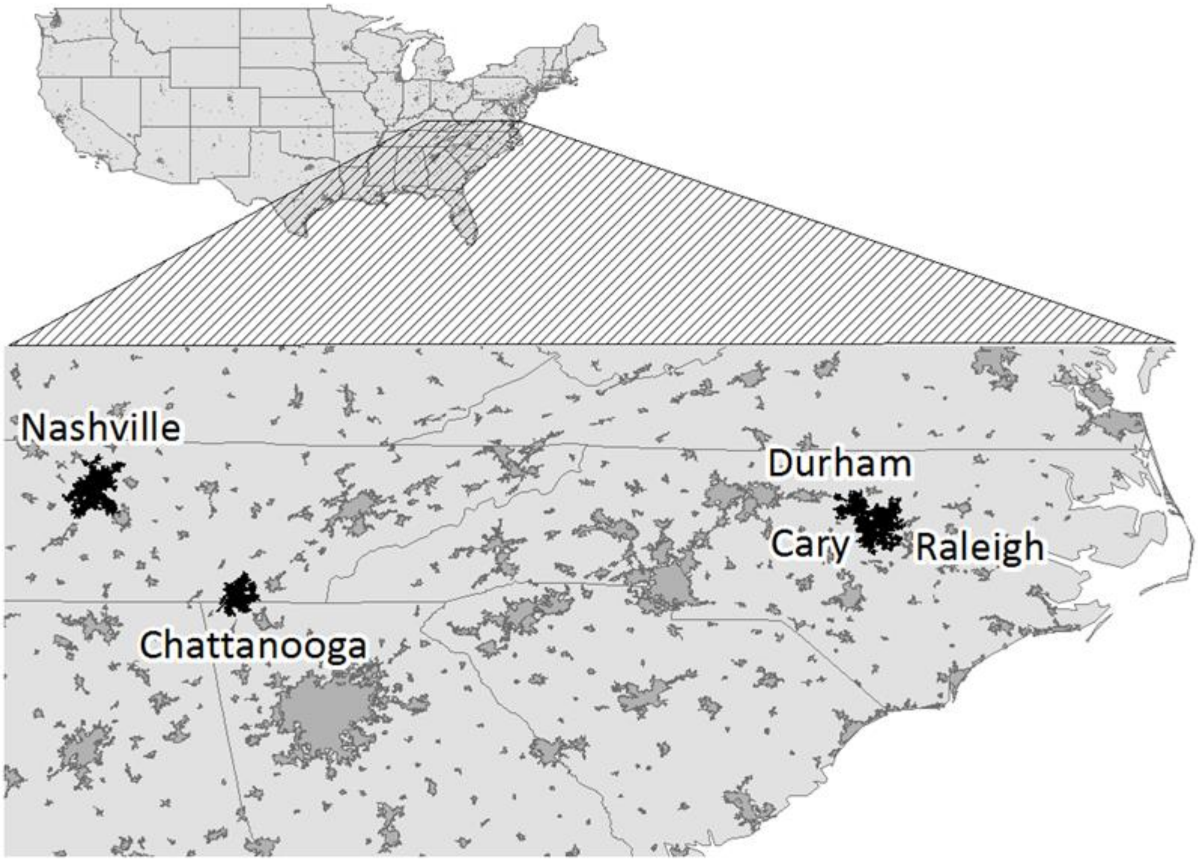

Figure 3.1: Locations of bioretention systems and extent of project area.

Table 3.1: Summary of observed bioretention sites.

City	Value	Surface Area (m^2)	Available Ponding Depth (cm)	Trees per practice	Number of Observed Sites	Avg. Annual Rainfall[a] (mm)
Chattanooga, TN	Minimum	27.6	0.0	1	8	1334
	Mean	298.3	21.1	4		
	Maximum	464.5	25.4	12		
Nashville, TN	Minimum	12.6	0.0	1	12	1201
	Mean	63.1	10.4	2		
	Maximum	120.0	22.9	4		
Raleigh, NC	Minimum	111.3	0.0	1	5	1100
	Mean	305.4	16.5	2		
	Maximum	571.7	35.6	5		
Cary, NC	Minimum	103.6	0.0	1	5	1176
	Mean	315.8	12.7	2		
	Maximum	523.0	38.1	3		
Durham, NC	Minimum	49.1	5.1	1	8	1130
	Mean	246.9	22.6	2		
	Maximum	912.2	27.9	4		

[a]Arguez et al., 2010.

healthy tree growth and condition, whereas smaller, patchy crowns indicate growth decline or poor tree health (Zarnoch et al., 2004). The definitions of these crown condition indicators, their measurement and rating scales, influencing factors, and other procedural remarks can be found in Schomaker et al. (2007). Crown condition indicator ratings for each tree were averaged between observers and recorded in the field. Measurements of tree height, diameter at breast height (DBH), root collar diameter (RCD), and height to crown base were collected. Sources of shade and scarring of the lower stem (mechanical damage, sun scald, freezing) were noted and photographed when present. For consistency, the same observers conducted crown condition ratings for all trees surveyed in this study.

Two composite crown indicators, composite crown volume (CCV) and composite crown surface area (CCSA), were calculated for each tree by approximating the shape of the crown as parabolic and using crown density ratings to estimate the portion of crown volume and/or surface area comprised of biomass (Zarnoch et al., 2004, Schomaker et al., 2007). Once CCV and CCSA were calculated, raw composite crown indicator values were directly compared with other individuals in the same species or standardized to a mean of 0 and a standard deviation of 1 for comparison across species (Zarnoch et al., 2004). The formulas used to calculate CCV and CCSA are shown in Equations (1) and (2) (Zarnoch et al., 2004). Further information on the calculation and use of composite crown indicators can be found in Zarnoch et al. (2004) and Schomaker et al. (2007).

$$CCV = 0.5\pi R^2 CL * CD \tag{1}$$

$$CCSA = \frac{4\pi CL}{3R^2}\left[\left(R^2 + \frac{R^4}{4CL^2}\right)^{1.5} - \left(\frac{R^4}{4CL^2}\right)^{1.5}\right] * CD \tag{2}$$

where: R = CDIA/2 (m)

CDIA = crown diameter (m)

H = tree height (m)

CL = H*(LCR)/100

LCR = live crown ratio

3.3.3 Soil Sample Collection

Soil samples were collected for particle size distribution and soil chemical analyses to characterize the subsurface growing conditions in each bioretention system. Prior to collecting samples, mulch and top soil layers were removed to expose the underlying bioretention media. Samples from the top 10–20cm of bioretention media were taken randomly throughout each system and composited to account for spatial variability. Coarse particle size distributions were conducted using procedures outlined in ASTM D422-63 Standard Test Method for Particle-Size Analysis of Soils (ASTM, 1998) to provide a general composition estimate of the bioretention media collected at each site (i.e., percentages of gravel, sand, and clay/fines). Soil samples were also sent to the Clemson Agricultural Service Laboratory for chemical analysis of several soil attributes (soil pH, buffer pH, extractable phosphorous, calcium, magnesium, zinc, manganese, copper, boron, and organic matter). Summary statistics of soil attributes selected based on modeling efforts discussed subsequently are shown in Table 3.2.

3.3.4 Comparison with Non-Bioretention Trees

Data collected during previous i-Tree Eco urban tree assessments from North Carolina State University (NCSU) (Raleigh, North Carolina), Georgia Tech University (Atlanta, Georgia), and the City of Atlanta, Georgia, were used to facilitate a health comparison between urban trees and trees planted in bioretention systems observed in this study (Kuehler, 2016, Blood et al.,

Table 3.2: Soil chemistry and particle size distribution results.

City	Value	Particle Size Distribution		Bioretention Media Analysis (Select Results)				
		Sand (%)	Fines (%)	pH	Organic Matter (%)	Cu (kg/ha)	K (kg/ha)	Buffer pH
Chattanooga, TN	Minimum	68.3	9.4	6.9	1.9	0.1	43.7	7.7
	Mean	74.9	23.6	7.3	2.4	4.3	74.8	7.8
	Maximum	87.2	31.7	7.8	4.2	8.9	134.5	7.8
Nashville, TN	Minimum	64.2	9.4	6.8	3.2	0.0	41.5	7.3
	Mean	76.7	16.7	7.5	6.5	0.2	106.8	7.6
	Maximum	81.9	23.8	7.8	14.3	0.7	230.9	7.8
Raleigh, NC	Minimum	58.1	16.0	6.1	2.1	1.0	58.3	7.6
	Mean	67.6	30.6	6.6	4.5	3.4	93.1	7.7
	Maximum	83.9	40.0	7.4	6.1	5.4	136.7	7.9
Cary, NC	Minimum	39.7	21.3	5.6	1.1	0.6	28.0	7.7
	Mean	51.2	48.5	6.3	1.8	1.2	97.6	7.8
	Maximum	78.7	60.3	6.8	2.7	1.9	180.5	7.9
Durham, NC	Minimum	65.3	16.3	5.7	0.6	0.4	26.9	7.7
	Mean	75.5	23.6	6.5	2.3	1.3	67.3	7.8
	Maximum	83.5	32.4	7.1	7.0	4.3	143.5	7.9

2016, Rudder, 2011). To maximize similarities between the populations, reference data taken from the i-Tree Eco inventories (referred to as non-bioretention trees) were filtered to include only trees from the same species and within the range of DBH values observed in the bioretention systems. Ranges in DBH were used for comparison because other parameters that may have better captured the similarities between trees, such as date of planting, tree age, and growth rate, were not available in the referenced data. The i-Tree data were further limited to include only trees grown in open areas (crown light exposure > 3) to mimic the open growth conditions observed in the bioretention systems. Composite crown indicators were calculated from this subset of i-Tree Eco data and compared with trees grown in bioretention systems to analyze differences in tree health. Comparisons between bioretention and non-bioretention trees were limited to the data available in the i-Tree databases. Therefore, information on soil conditions at these sites, which may have provided further insights into the differences in growing conditions between the bioretention and non-bioretention trees, was not available for this study. To fully illustrate trends in species health, maximize the number of potential reference trees in the non-bioretention i-Tree Eco data, and allow adequate statistical power in analyses, comparisons of bioretention and non-bioretention tree health were limited to the six most frequently observed species identified in bioretention systems (Table 3.3).

3.3.5 Random Forest Modeling

Numerical models of the influence of environmental factors on tree health and their relative importance were conducted using random forest (RF) regression analyses (Breiman, 2001). Random forest, which can be used for classification or regression, is an ensemble learning method that aggregates the results from a large number decision trees to model a possibly

Table 3.3: DBH range and location of tree species in bioretention systems.

Species	DBH (cm)		Chattanooga, TN	Nashville, TN	Raleigh, NC	Cary, NC	Durham, NC	Tot.
	Min-Max	Med						
Red Maple	6.9-48.5	14.5	6	0	0	2	5	13
River Birch	2.8-127.8	8.9	15	1	2	4	3	25
Lacebark Elm	6.9-53.8	29.0	0	15	0	0	0	15
Bald Cypress	8.4-26.4	16.5	0	4	3	1	1	9
Redbud	3.6-32.3	13.5	6	2	0	0	0	8
Pin Oak	8.6-14.0	9.7	0	1	3	0	1	5
Other	-		4	5	4	1	8	22
Total	-		31	28	12	8	18	97

nonlinear relationship between predictor and response variables in a data set (Liaw and Wiener, 2002). In the case of regression (as used in this study), bootstrap samples from the data set are used to construct regression trees such that, at each decision node, a best split is determined using a randomly selected subset of the predictor variables. The average of outputs by all trees is reported as the estimated response variable (Cartus et al., 2012). Estimations of error rates are obtained by predicting data not included in the bootstrap sample (out-of-bag, or OOB, data) and averaging the resulting errors across all trees (Liaw and Wiener, 2002). Variable importance can be evaluated by randomly reordering an OOB array of a predictor variable while keeping all others constant and measuring the percentage increase in mean square error (MSE) in the response variable prediction compared with the original result (Liaw and Wiener, 2002).

The random forest regression method has several advantages that were considered when selecting a modeling strategy for this study. Due to the randomness associated with selecting bootstrap samples and predictor variables used at each decision node, random forests can account for the complex interactions of predictor variables (e.g., soil nutrients) and their resulting impact on response variables (CCV and CCSA). Next, because of the large number of regression trees that are generated, overfitting of the model to the original data set is generally avoided (Breiman, 2001). Finally, the measure of variable importance produced by the models provides an interpretable metric with which to compare various predictor variables and their relative influence on a response variable. Since their inception, random forest models have been used in a number of fields, with forestry-specific applications ranging from predicting ecological responses to climate change scenarios (Prasad et al., 2006) to remote collection of stand-level canopy height and growing stock volume of forest plantations in Chile (Cartus et al., 2012).

The results of two random forest regression models developed to predict CCV and CCSA values were used to determine the importance and influence of several observed and measured bioretention site parameters on tree health. Because the composite crown indicators from various species were combined in the random forest analyses, CCV and CCSA values for each species type were standardized to a mean of 0 and a standard deviation of 1 to provide a more meaningful comparison across different species (Zarnoch et al., 2004). Hardwood species represented by fewer than five observations were grouped as “Other hardwoods,” after which their individual composite crown indicators were standardized across the group. Measurements of bioretention soil (particle size distribution, soil chemistry), bioretention design factors (surface area, available ponding depth), and general site conditions were used as predictor variables in both random forest models. Refer to Table 3.4 for the 19 predictor variables used to initiate each model to predict CCV and CCSA values for all 97 bioretention trees that made up the data set.

The default number of decision trees created in the randomForest package is set to 500; however, depending on the situation, increasing the number of trees may be necessary to stabilize prediction estimates and optimize model error (Liaw and Wiener, 2002). Due to the relatively small size of the data set used in this study compared with others (such as Prasad et al., 2006), the number of regression trees created in each random forest was increased to 25,000. This selection was made after testing the prediction accuracy of various sizes of regression random forests and weighing the required computing time with incremental increases in model accuracy. A recursive variable elimination procedure was implemented in the random forest models to reduce the influence of less-important “noise” variables on model performance. Errors in model performance (i.e., the difference between the estimated response variable and the

Table 3.4: List of predictor variables used to initialize random forest models.

Variable	Description
Species	Tree species planted in bioretention
Surface Area	Surface area practice (ft^2)
Ponding Depth	Max. available ponding depth (in)
Percent Sand	Percent passing sieve no. 16 (%)
Percent Fines	Percent passing sieve no. 50 (%)
Tree Location	Tree planting location (upslope/midslope/bottom)
Shading	Potential for external shading (Y/N)
SpH	Soil pH
P	Soil phosphorous (lb/acre)
OM	Soil organic matter (%)
K	Soil potassium (lb/acre)
Mg	Soil magnesium (lb/acre)
Zn	Soil zinc (lb/acre)
Mn	Soil manganese (lb/acre)
Cu	Soil copper (lb/acre)
B	Soil boron (lb/acre)
Na	Soil sodium (lb/acre)
BpH	Soil buffer pH
Ca	Soil calcium (lb/acre)

measured CCV and CCSA values) were objectively evaluated using leave-one-out cross validation (Kohavi, 1995). To account for variations in performance, the model was executed 25 times. In each iteration of recursive variable elimination, the predictor variable with the lowest importance is eliminated. Therefore, the iteration number in which a predictor variable was eliminated was used as a measure of its relative importance to the response variable.

3.3.6 Statistical Analysis

All statistical analyses were completed using R statistical software (R Core Team, 2016). Tests for normality indicated that the data were not normally distributed, necessitating the use of nonparametric analyses. Thus, Wilcoxon rank sum tests were performed to determine whether the data suggested a statistically significant difference in CCV and CCSA between bioretention and non-bioretention trees (Ott and Longnecker, 2010). Random forest regression analyses of CCV and CCSA were conducted using the randomForest package developed by Liaw and Wiener (2002).

3.4 Results and Discussion

3.4.1 Tree Species Identified in Bioretention Systems

In total, 97 trees in bioretention systems were observed during the field study. Over 20 species were identified, ranging from nonnative ornamental species such as Kwanzan cherry (*Prunus serrulata*) to native species common to the southeastern United States, such as red maple (*Acer rubrum*), river birch (*Betula nigra*), and bald cypress (*Taxodium distichum*). The six species identified in Table 3.3 were the most frequently observed trees in bioretention systems and accounted for over 75% of the trees analyzed in the field study.

Many of these species are recommended selections for urban areas in the eastern United States due to characteristics such as large mature size, ability to tolerate a wide range of soil moisture and pH conditions, and, in some instances, seasonal aesthetics (autumn foliage and/or spring flowers) (Bassuk et al., 2009). Outside of urban stormwater management, other research has identified roles that a number of these species play in environmental remediation and landscape management systems. Examples of such studies include phytoremediation of groundwater contaminants using bald cypress trees (Fontenot et al. 2014), streambank stabilization using river birch plantings (Simon and Collison, 2002), and reestablishment of red maple trees on abandoned surface mining lands (Evans et al., 2013). Overall, the species most often represented in bioretention systems are logical considering existing design guidance and species traits that make them viable urban tree selections.

3.4.2 Comparison of Bioretention and Non-Bioretention Tree Health

After filtering the non-bioretention data to match the species and DBH ranges measured in the field (Table 3.3), the compiled i-Tree Eco data provided 985 urban trees for comparison (Table 3.5) (Kuehler, 2016, Blood et al., 2016, Rudder, 2011). As with the bioretention tree data, the non-bioretention data were used to calculate CCV and CCSA values using Equations (1) and (2). Because tree health comparisons were made within the same species, raw CCV and CCSA values could be used without any statistical transformation (Zarnoch et al., 2004). Figures 3.2 and 3.3 show box plots of CCV and CCSA values for bioretention and non-bioretention trees, along with species-specific Wilcoxon rank sum results.

In conjunction with the box plots, the Wilcoxon rank sum test results indicate that, except for bald cypress (*Taxodium distichum*), CCV and CCSA values of bioretention trees are less than

Table 3.5: Non-bioretention tree data compiled from i-Tree Eco inventories.

Species	Median DBH[a] (cm)	NCSU	Georgia Tech	City of Atlanta	Total
Red Maple	20.3	65	327	33	426
River Birch	21.1	54	99	8	161
Lacebark Elm	17.0	24	161	8	193
Bald Cypress	13.2	1	45	0	46
Redbud	8.1	51	49	3	103
Pin Oak	9.7	5	51	0	56
Total	-	200	733	52	985

[a] Range of DBH values limited to measured DBH values of bioretention trees (see Table 3.3).

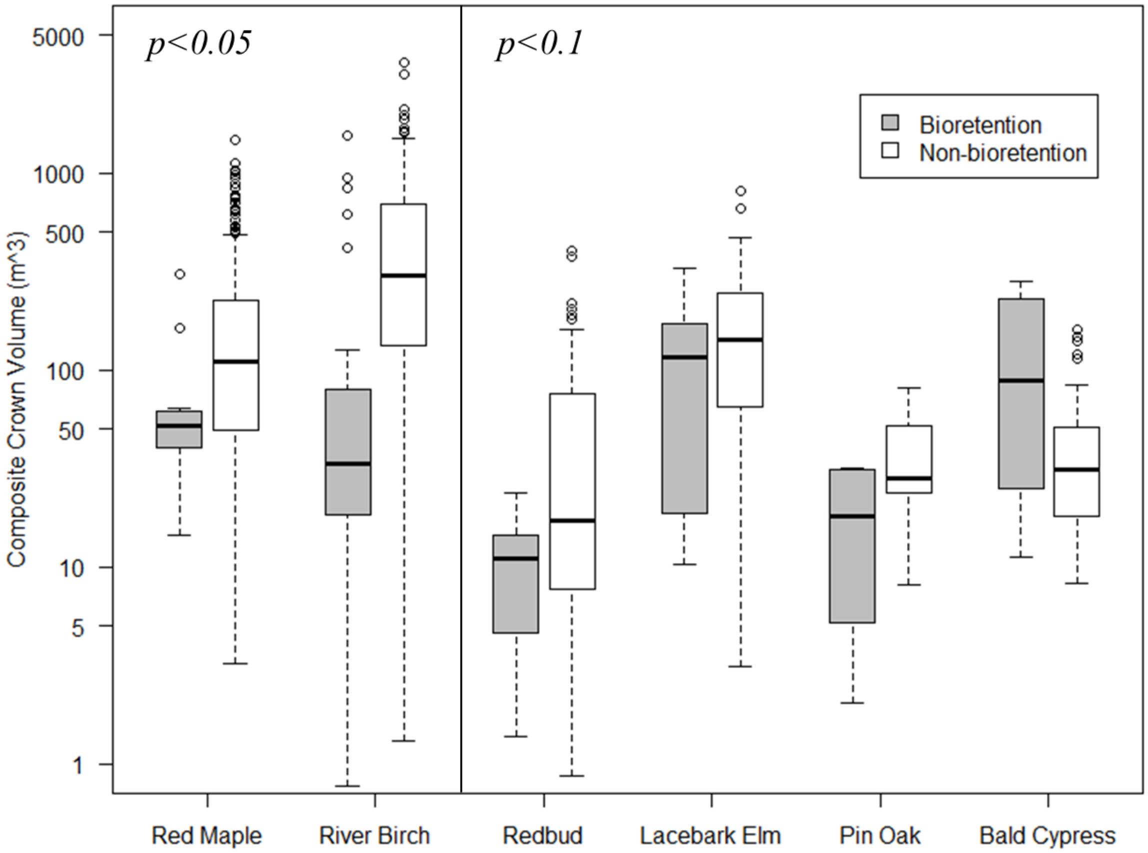

Figure 3.2: CCV (m^3) comparison of species in bioretention practices to non-bioretention trees of the same species, growing condition, and DBH range. Differences between composite crown volumes of bioretention and non-bioretention red maple and river birch were significant at $p<0.05$, while all other species were significant at $p<0.1$.

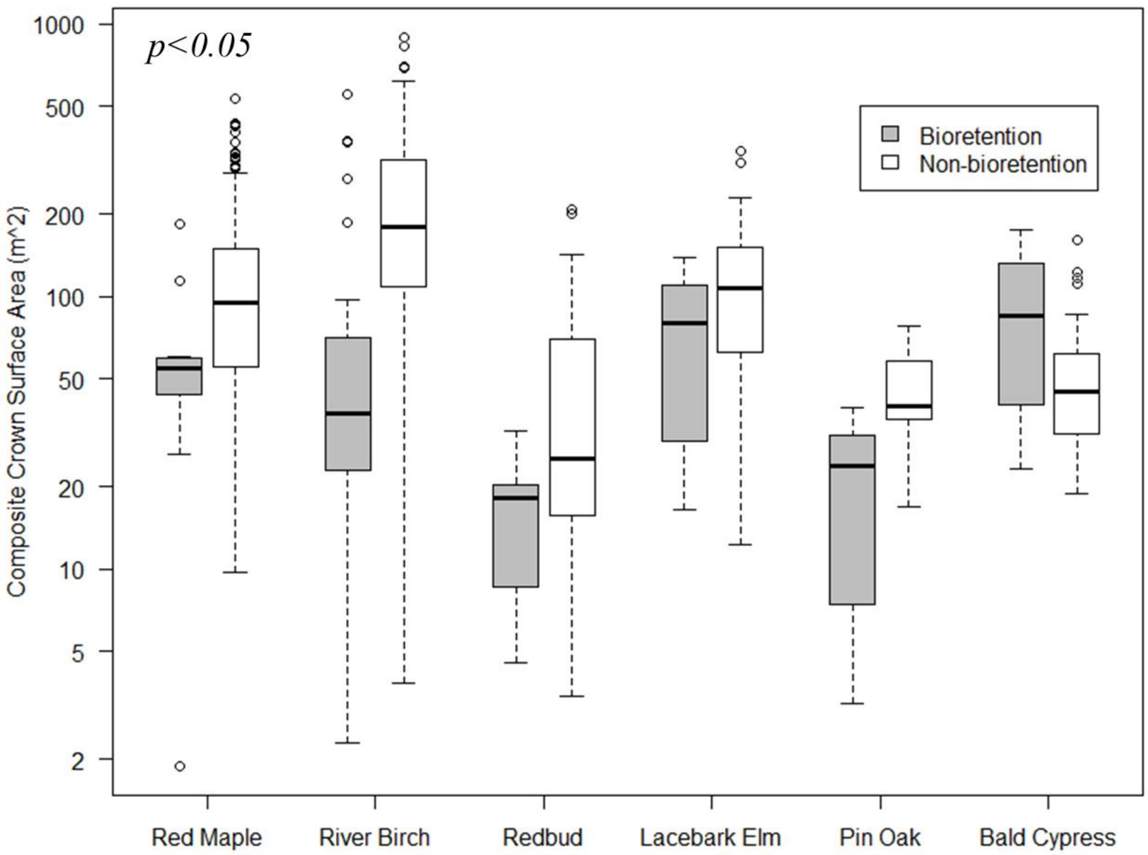

Figure 3.3: CCSA (m^2) comparison of species in bioretention practices to non-bioretention trees of same species type, growing condition, and DBH range. Differences between composite crown surface areas of bioretention and non-bioretention trees were significant at $p<0.05$ for all species.

those of non-bioretention trees. CCVs of red maple (*Acer rubrum*) and river birch (*Betula nigra*) planted in bioretention systems were less than CCVs of similar urban trees planted outside of bioretention systems ($p<0.05$). CCV and CCSA values for eastern redbud (*Cercis canadensis*), lacebark elm (*Ulmus parvifolia*), and pin oak (*Quercus palustris*) trees in bioretention were less than non-bioretention trees of the same species and DBH range at $p<0.1$ and $p<0.05$, respectively. Unlike the other species, bald cypress trees had significantly larger CCV ($p<0.1$) and CCSA ($p<0.05$) values in bioretention than in non-bioretention. These trends suggest that many of these tree species are in a less healthy state when grown in bioretention compared with those in non-bioretention, whereas only bald cypress trees appear to be healthier in bioretention.

The results demonstrate the influence of species-specific site preferences on the health of trees in bioretention systems and highlight the need for their consideration when selecting species during bioretention design. Tree tolerance for a number of environmental elements, such as soil type, sun exposure, soil moisture regime, soil pH, and climate, can differ widely between species. Thus, tree species selection for a given project that does not consider these components can result in poor tree health and diminished benefits derived from urban trees (Bassuk et al., 2009). Although not included in the scope of this work, it should be noted that tree crown condition may also be influenced by an array of external factors after establishment, including limb removal, fertilizer application, storm damage, disease, pests, mechanical damage, and extreme weather conditions (e.g., prolonged or severe drought). The remainder of the discussion in this section focuses on tree species suitability for bioretention systems based on soil and environmental preferences, which can be considered during design to select an appropriate tree species.

Adapted from information originally presented in Bassuk et al. (2009), Figure 3.4 shows the soil pH and range of soil moisture preferences (represented by the shaded regions of the figure) of the six bioretention tree species. To varying degrees, environmental conditions expected in bioretention systems (well-drained soils characterized by periods of both soil dryness and inundation following rain events) fit within the tolerance ranges for each of the six species. However, the soil pH values of the media presented in Table 3.2 were at the highest end of, or slightly above, the preferred ranges for these species. Despite the overlapping of soil moisture and pH preferences common to many of the species, inferences can be formulated from this information to explain the health differences of these species when planted in bioretention and non-bioretention settings. For example, the neutral to slightly alkaline soils found in bioretention systems may have negatively impacted some species more than others (Table 3.2), and the dynamic soil moisture conditions characteristic of sandy bioretention soils may have been outside the narrower preferences of species such as red maple, potentially impacting tree health. Similarly, periods of inundation that resulted from influent stormwater runoff may have been detrimental to the health of species preferring drier soil conditions, such as lacebark elm and eastern redbud.

Consulting established knowledge of the silvics of these species, especially with respect to preferred soil type and natural habitat, can shed further light on the suitability of a species for use in bioretention systems. Dickson (1990) reported that while eastern redbud trees tolerate nutrient-deficient environments and can be found in a variety of soil textures, they are not found in soils comprised of coarse sand. Likewise, a study of ecological factors influencing river birch trees in North Carolina found that soils in river birch stands are characterized by significantly

Species	Soil pH	Saturated or very wet soil	Moist, well-drained soil	Occasionally dry soil	Very dry soil
Bald Cypress	4.5-6.0				
Pin Oak	4.5-6.5				
River Birch	3.0-6.5				
Red Maple	4.7-7.3				
Redbud	5.0-7.9				
Lacebark Elm	4.8-7.0				

Figure 3.4: Species-specific site preferences (adapted from Bassuk et al., 2009; soil pH data from USDA and NRCS, 2017).

higher clay and soil organic matter content than non-river birch stands, suggesting that river birch trees can tolerate (and may even require) high soil moisture environments on a year-round basis (Wolfe and Pittillo, 1977). Similarly, the natural habitat conditions of drought intolerant pin oak trees are characterized by heavy-textured, poorly drained soils that can result in prolonged seasonal surface flooding (Fowells, 1965, Sullivan and Levitt, 1959). As previously discussed, a typical bioretention environment is not representative of many of these natural conditions, deviations from which could be expected to negatively impact tree condition. Overall, these studies suggest that the lower state of health found in red maple, river birch, lacebark elm, and eastern redbud trees planted in bioretention systems can be attributed to the lack of compatibility between the natural habitats of these species and the bioretention environment. Further, while soil data were unavailable for the non-bioretention tree locations, the poorer health of these trees planted in bioretention systems suggest that, for these species, one or more environmental variables found in the bioretention areas did not meet the biological requirements for growth to the same extent as do native growing conditions. It should be noted that in locations where in-situ soils are predominately sandy, the results of a similar study may vary as conditions within and outside of the bioretention cell may be more similar.

Unlike the species exhibiting poorer health in bioretention relative to non-bioretention urban trees, the improved state of health observed for bald cypress trees planted in bioretention suggests that the biological needs of this species are met by these growth conditions. As this species is outside of its natural range in the area surveyed, planted urban trees may not be well matched to species preferences, whereas the environment of the bioretention area more closely approximates conditions in its natural range. Bald cypress trees prefer slightly acidic to somewhat neutral soils and a wide range of moisture conditions ranging from periods of occasionally dry to fully saturated, all of which are common to bioretention systems (Figure 3.4). Additionally, the best growth of bald cypress has been reported to occur on moist, fine sandy loam soils with moderately good drainage, although they are more commonly found in very wet, clayey soils in low-lying areas because of competition with higher-tolerance hardwoods (Fowells, 1965). This suggests that the environment in a typical bioretention system is a close analog to the preferred natural growing conditions of bald cypress trees and supports the conclusion that they were in a healthier state when planted in bioretention systems compared with non-bioretention settings. This result also demonstrates the need for an analysis of species suitability for bioretention systems on the basis of environmental tolerances and preferred natural growing conditions prior to tree selection and planting. Within a species, the characterization and selection of cultivars or ecotypes tolerant of the bioretention environment may also lead to improvements in tree health. Further, because of the large number of variables that can influence tree distribution and health, it is vital to identify factors that might be most influential to trees in these systems.

3.4.3 Environmental Influences on Bioretention Tree Health

The average errors observed over the 25 executions of the random forest models of CCV and CCSA for any given number of parameters used in the models are presented in Figure 3.5. The trends in performance suggest that errors in the random forest regression models were, on average, minimized when eight and six predictor variables were used to predict CCV and CCSA, respectively. For models of CCV using eight predictor variables, the average error across the 25 model executions was approximately 59.8, or about 0.62 standard deviation units away from the measured CCV values for each of the 97 trees used in the model. Similarly, for models of CCSA using six predictor variables, the average error across all 25 executions was approximately 58.7, or about 0.61 standard deviation units away from the observed CCSA values for each bioretention tree.

Table 3.6 presents the ordered lists of predictor variables based on their importance in relation to the response variables CCV and CCSA averaged across the 25 model executions. The ordering of predictor variable importance in relation to the response variables was generally consistent for both CCV and CCSA. Further, the results indicate that a particular subset of the predictor variables had an important influence on CCV and CCSA, and thus tree health, in the bioretention systems. Specifically, a comparison of the results in Figure 3.5 and Table 3.6 suggests that a particular subset of eight of the predictor variables (variables that minimized the error rates in both CCV and CCSA models) are most important to tree health [organic matter (OM), percentage fines, percentage sand, buffer pH (BpH), potassium (K), species, copper (Cu), and tree location]. While each of the predictor variables may influence tree health to an extent and should be considered as needed, the consistent ranking of the most important variables suggests that an emphasis should be placed on characterizing these eight parameters and investigating their suitability for various species prior to selecting trees to include in bioretention.

Generally, the eight most-important predictor variables can be categorized into three groups, which are summarized in Table 3.7: (1) bioretention media composition (percentage fines, percentage sand, OM); (2) bioretention media chemistry (BpH, Cu, K); and (3) tree selection/planting (species, tree location).The predictor variables in the bioretention media

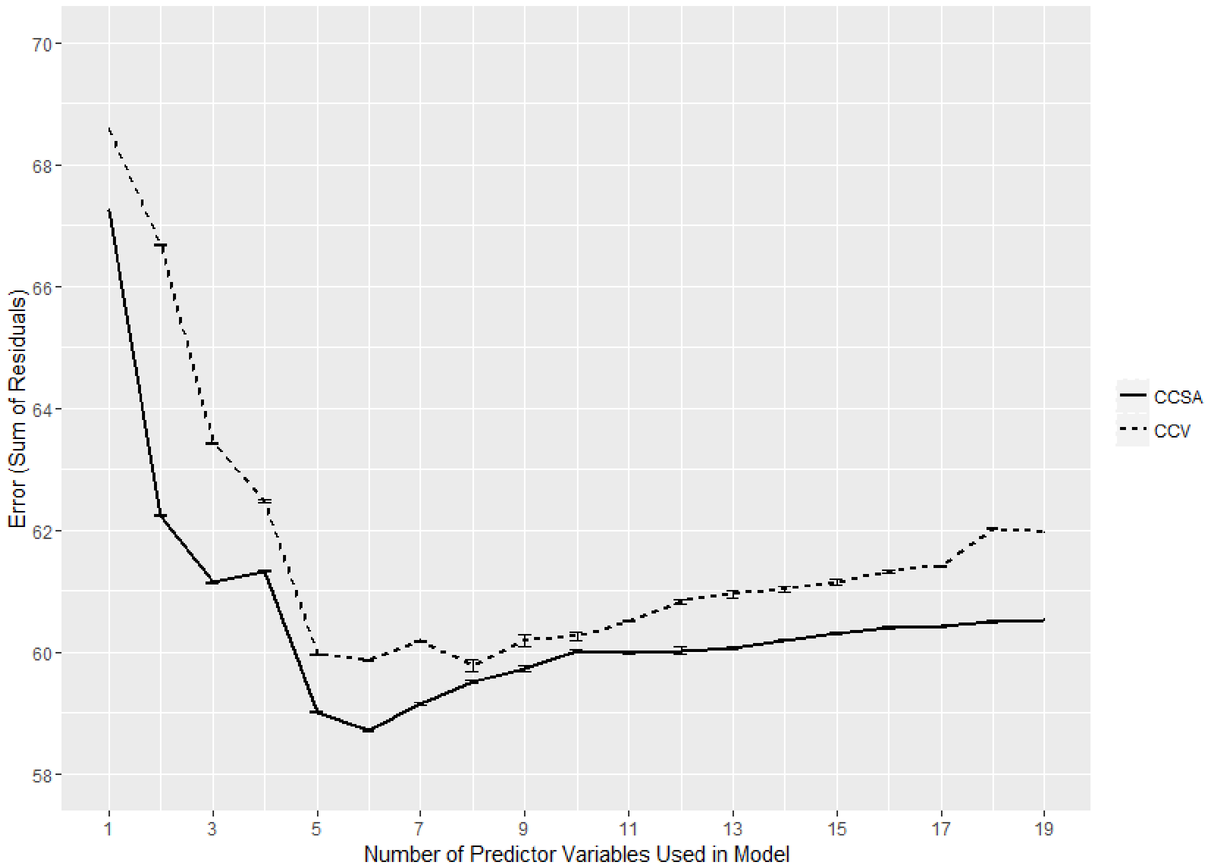

Figure 3.5: Random forest model error values averaged over 25 executions for the response variables CCV and CCSA. Error bars indicate 95% confidence intervals on the mean error.

Table 3.6: Random forest results for predictor variable importance rankings.

Predictor Variable	Average Rank	
	CCV	CCSA
OM	**1**	**1**
BpH	**2**	**6**
Percent Fines	**3**	**2**
Percent Sand	**4**	**3**
Species	**5**	**5**
K	**6**	**4**
Cu	**7**	8
Tree Location	**8**	17
Ca	9	10
P	10	9
SpH	11	7
Mn	12	14
Zn	13	16
Mg	14	15
Na	15	13
Ponding Depth	16	11
B	17	18
Surface Area	18	12
Shading	19	19

Note: rank 1 = most important. Bold indicates predictor variables were used in random forest models with lowest error.

composition category support the conclusions from the comparison of tree health in bioretention and non-bioretention urban settings. Media composition and its alignment (or lack thereof) with a species' preferred native growing habitat was determined to be one of the factors contributing to health discrepancies between bioretention and non-bioretention trees. This finding is reinforced by the high-importance ranking of the percent fines and percent sand parameters in the random forest models of CCV and CCSA. Soil OM enhances soil structure and water retention, promotes biological function in soils, and acts as a reservoir of several key nutrients for plants and microorganisms. Research has associated high OM in bioretention soils with nutrient export; thus, most bioretention media are typically very low in OM to ensure adequate sequestration of nutrients in stormwater (Hunt et al., 2012). However, it is important to note that OM content specifications for bioretention media are highly variable in bioretention design standards on a state-by-state basis, ranging from 3%–5% in North Carolina (NCDEQ, 2009) to 15%–25% in Minnesota (MPCA, 2016), and that nutrient contributions from OM can vary depending on the type used in the bioretention media. Because organic matter is linked to soil fertility, the high importance ranking of soil OM further emphasizes the considerations that must be part of selecting OM content that supports tree health as well as the ability of a tree species to tolerate the nutrient deficient environments found in low-OM bioretention media.

The high-importance soil parameters in the bioretention media chemistry category (BpH, Cu, K) emphasize the need for bioretention media analysis and chemical characterization prior to selection of trees for bioretention systems. Buffer pH relates to the ability of a soil to resist changes in pH, which is an important soil parameter for many tree species with narrow tolerance ranges of soil pH (Figure 3.4) and can be an indicator of how soil chemistry and nutrient

Table 3.7: High importance predictor variables and influences on tree health.

Category	Predictor Variable	Comments
Bioretention Media Composition	Fines (%)	Impacts soil moisture dynamics; bioretention media should align with species-specific habitat preferences
	Sand (%)	
	Organic Matter (%)	Influences soil fertility, structure; OM standards vary between regulatory agencies
Bioretention Media Chemistry	Buffer pH	Reflects possible changes in bioretention media pH and soil chemistry over time
	Copper	Used by trees as a micronutrient; deficiency leads to crown defoliation and dieback
	Potassium	Vital to plant functions (photosynthesis, water regulation, cell expansion); required by trees in large amounts for healthy growth
Tree Selection and Planting	Planting Location	Tree planting location within the bioretention practice (upslope, bottom of bowl, etc.) should reflect tree tolerance to inundation
	Species Selection	Species selected should be tolerant of unique bioretention environment

availability change over time in bioretention. Copper (Cu) is used by trees as a mineral micronutrient and, though only required in small amounts, is found in key enzymes including ascorbic acid oxidase (Pallardy, 2008). Excessive Cu levels can be toxic to trees, whereas deficiencies can lead to defoliation and crown dieback—two influential factors in composite crown indicators and tree health (Pallardy, 2008). Interestingly, all of the bioretention media samples collected would be considered extremely deficient in Cu compared with typical natural growing environments, which normally range from 25–200ppm [approximately 56–448 kg/ha (50–400 lb/acre)], likely attributable to the high soil pH values of the bioretention media (Table 3.2) (Stone, 1968). It should be noted that while Cu was identified as a high-importance parameter in the data set, potential deficiencies in other micronutrients, including iron, zinc, molybdenum, and chlorine, which would also influence tree health, should be investigated and identified prior to species selection. Potassium is required in larger amounts and is used in a variety of vital plant functions, including opening and closing of stomata, photosynthesis reactions, and cell expansion and growth (Pallardy, 2008). Because of its role in regulating water flow in the plant, its availability may be of particular importance where trees must tolerate large fluctuations in soil water. Potassium is commonly added via fertilizers to achieve sufficient nutrient levels in soils, a strategy that may need to be implemented in nutrient-deficient bioretention media to maintain tree health, although the potential for nutrient export should be considered prior to making any soil amendments.

Characterization of the six predictor variables in the bioretention media composition and bioretention media chemistry categories can address the remaining high-importance parameters, species and tree location (tree species/planting category). Species selection should be guided by findings from the comparison of tree health, which indicate that species with native growing habitats that match the conditions found in bioretention systems (sandy, well-drained, low-OM soils with fluctuating periods of soil moisture conditions) exhibit improved health compared with non-bioretention trees. When an appropriate species is ultimately identified, the final location of the tree in the system (tree location) should be carefully selected prior to planting. Trees planted in the bottom of the bioretention system may be subjected to longer periods of inundation as the system fills with water, which can inhibit water absorption and nutrient uptake in species that are not adapted to saturated soil conditions (Pallardy, 2008). Conversely, trees planted in the upper slopes of the system may be subjected to compacted subsoils underlying surrounding impervious areas, which can limit soil moisture and aeration and impact growth and overall tree health (Pallardy, 2008). Some tree species, such as river birch, which requires moisture yet is intolerant of inundation, may be suitable only for midslope positions. Selection of an appropriate species informed by analysis of the chemical and physical composition of the bioretention media and planting location may be a critical step in promoting high functioning, healthy trees in bioretention systems.

3.5 Conclusions and Recommendations

Trees in bioretention systems in Tennessee and North Carolina were studied to characterize their health status and identify which environmental factors are most influential to their health and overall function. Tree health was first investigated by comparing bioretention trees to analogous non-bioretention urban trees based on two three-dimensional composite crown indicators, CCV and CCSA. Results from this comparison showed that trees from five of the six species examined in the study [red maple (*Acer rubrum*), river birch (*Betula nigra*), pin oak

(*Quercus palustris*), eastern redbud (*Cercis canadensis*), and lacebark elm (*Ulmus parvifolia*)] were less healthy (smaller CCV and CCSA values) than similar non-bioretention trees, whereas only bald cypress (*Taxodium distichum*) exhibited better health in bioretention. This outcome was linked to differences (or similarities) between species-specific preferences for site condition, such as soil type/composition, soil moisture, and buffer pH, and conditions expected to be found in bioretention systems (sandy, well-drained, nutrient-deficient soils with frequent periods of inundation/drought). These results suggest that natural growing habitats and species-specific preferences for site condition should be considered when selecting a tree species for a bioretention practice.

The relative importance of a number of bioretention parameters and their influence on tree health was modeled using random forest regression models of CCV and CCSA. Results from analyses of soil particle size composition and soil chemistry, along with observations of species type, planting location, and various bioretention system characteristics, were used as predictor variables in each model. It was determined that the random forest regression models exhibited the lowest error levels when eight and six predictor variables were used to model CCV and CCSA, respectively. These predictor variables can be categorized into three groups—bioretention media composition (percentage fines, percentage sand, organic matter), bioretention media chemistry (buffer pH, copper, potassium), and tree species/planting location—which should be prioritized when selecting tree species to include in bioretention systems.

Based on the results of this study, the following design recommendations can be implemented to promote tree health and function in bioretention systems:

- Tree species should be selected based on their ability to tolerate the unique growing conditions found in bioretention systems along with the stresses associated with urban environments. Considerations of species-specific preferences for site conditions, such as soil type/composition, soil moisture, soil pH, and nutrient availability, and comparisons of the natural growing conditions of a species with the likely conditions of the bioretention environment should guide tree species selection for bioretention systems.
- Bioretention media composition should be characterized through particle size distribution and soil chemical analysis, with priority placed on investigating the high-importance soil parameters identified in this study (organic matter, percentage fines, percentage sand, buffer pH, potassium, and copper and other micronutrients). Tree species selection should be informed by the results of these analyses and tree locations should be optimized for a particular species prior to planting. If not already specified, these parameters should be added to bioretention media specifications as guidance for future projects.

Research on tree health in bioretention systems should seek to expand the number and diversity of tree species as well as the geographic range used in this study. There are possible location effects on tree health that could not be analyzed herein due to variations in species representation between cities that should be explored in future work. Increasing both the number of trees and the number of species in the regression models of CCV and CCSA could replace the "Other hardwoods" category used in the random forest regression models. Incorporating potentially influential parameters that were not available in this study, such as catchment size and imperviousness, in models may identify additional (or alternative) bioretention parameters that influence tree health. Future research should also focus on trees in bioretention throughout their lifespan, comparing various planting techniques and monitoring the growth rate of bioretention trees compared with other urban trees to investigate how the unique characteristics of bioretention systems influence tree growth after planting. Studies should examine the differences in nutrient and water availability between bioretention media and other urban soils, which are typically characterized by low organic content, foreign materials, and a high degree of compaction, to investigate the influence of soil composition and quality on tree health in bioretention systems. Finally, expanding the area of study, which was limited to the inland southeastern United States, to include coastal or northern urban areas may influence the results of the bioretention–non-bioretention comparison of tree health, as the effect of both changing climates and native underlying soil conditions that more closely resemble the conditions found in bioretention may have a range of influences on the health of various tree species planted in bioretention systems.

3.6 References

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CHAPTER 4 : INVESTIGATING THE HYDROLOGIC AND WATER QUALITY PERFORMANCE OF TREES IN BIORETENTION MESOCOSMS

4.1 Abstract

Cities across the world are increasingly utilizing green infrastructure practices as part of their stormwater management programs. Bioretention areas have become a popular green infrastructure practice due to their widespread success in improving water quality and reducing runoff generated from impervious surfaces. Several studies have demonstrated that pollutant removal performance can be improved when plants are included in bioretention design; however, while numerous benefits of trees in urban areas have been identified, little knowledge of their contributions to stormwater management in green infrastructure currently exists. To address this need, a controlled mesocosm experiment was conducted to characterize the degree of stormwater treatment provided by bioretention columns planted with one of three native tree species commonly found across the eastern United States (*Acer rubrum* – red maple, *Pinus taeda* – loblolly pine, and *Quercus palustris* – pin oak). Tree pollutant removal performance was compared to nonvegetated mesocosms using a semi-synthetic stormwater mixture applied to the mesocosms over a period of 14 weeks. The hydrologic benefits of each species were characterized using data-logging scales placed below the mesocosms to compare evapotranspiration (ET) rates and drainage in each configuration. Differences in pollutant removal between tree species were largely not significant, indicating the dominant role of the bioretention media in mitigating dissolved and particle-bound constituents. Mesocosms planted with red maple (*Acer rubrum*) had significantly greater average ET rates (3.2 mm d^{-1}) than all other configurations, attributable to plant development and increased growth and canopy size. All mesocosms planted with trees had significantly higher ET rates than the nonvegetated mesocosms, illustrating the role of transpiration in bioretention hydrology which, depending on

species, accounted for 8.2-37.5% of average daily water losses from the mesocosms during testing. These results suggest that trees contribute to bioretention hydrology through evapotranspiration and that significant differences between species exist and are likely related to growth rate.

4.2 Introduction

Urban stormwater runoff is a significant contributor to impaired water quality and declining aquatic habitats in urban ecosystems throughout the world (USEPA, 1999a). Stormwater runoff from urban areas, which are characterized by concentrated human activity and widespread land use conversion to impervious cover, can contain several pollutants introduced through anthropogenic activity, such as nutrients, metals, hydrocarbons, bacteria, and suspended sediments derived from exposed or compromised urban soils (Hunt et al., 2012). As a result, polluted urban stormwater runoff can lead to eutrophication, deteriorated riparian zones, waterway closures, and reduced fishing, recreational, and aesthetic value of downstream waters (Hunt et al., 2006). Increasingly, cities and municipalities are implementing green infrastructure stormwater control measures (SCMs) and low impact development (LID) designs into their stormwater management programs. Because of their versatile design and established performance, bioretention practices have become one of the most popular SCMs implemented to manage urban stormwater runoff in cities worldwide (Davis et al., 2009).

Bioretention practices rely on both plant and soil processes to remove pollutants from urban stormwater runoff (USEPA 1999b). Though many design configurations have been employed, bioretention practices typically consist of an engineered sandy soil media topped with mulch and various plants. Stormwater runoff entering a bioretention system from the contributing drainage area, such as a parking lot, roadway, or other impervious area, slowly infiltrates through the sandy media, where pollutants are filtered out, adsorbed to soil particles, or taken up by plants or microbes prior to exiting the system. Numerous studies have documented the ability of bioretention practices to mitigate the hydrologic (e.g., Davis, 2008, DeBusk and Wynn, 2011, Winston et al., 2016) and water quality (e.g., Davis, 2007, Hatt et al., 2009, Li and Davis, 2009, Brown and Hunt, 2011) impacts of urban runoff on receiving waterbodies. Research has also shown that plants play a key role in these processes and enhance the performance of bioretention practices.

Several mesocosm-scale studies have characterized the pollutant removal contributions of plants in bioretention practices, though the plant types and species that have been investigated are limited. Many have observed that plants improved removal of nitrogen (N) and phosphorous (P), and differences between species and plant types have been reported (Lucas and Greenway, 2008, Read et al., 2008, Bratieres et al., 2008). Plant size, species selection, and root mass have been found to significantly influence plant contributions, and several studies have recommended a variety of grasses (including *Carex appressa*), shrubs, and rushes as preferential selections for bioretention practices (Read et al., 2008, Bratieres et al., 2008). In one of the few studies investigating the pollutant removal contributions of trees in bioretention practices, Denman et al. (2016) found that the presence of trees significantly improved soluble P and NO_x removal, though significant differences between species and soil types were not consistent and varied seasonally. The authors concluded that while trees reduced NO_x and P relative to unplanted controls, species selection did not influence nutrient removal performance (Denman et al., 2016).

Plants have also been shown to influence the hydrology of bioretention practices through evapotranspiration (ET). Several studies have used weighing lysimeters to record ET from bioretention practices planted with grasses, shrubs, and perennials. Wadzuk et al. (2015) found ET comprised up to 78% of the water budget of bioretention mesocosms planted with grasses. Similarly, Hess et al. (2017) used weighing lysimeters in rain gardens planted with switch grass, perennials, and deciduous shrubs in three media types with varied drainage configurations to conclude that ET accounted for between 43% and 70% of water losses. Scharenbroch et al. (2016) found that tree transpiration levels varied between species and accounted for 46% to 72% of the water balance from a parking lot in Illinois outfitted with green infrastructure practices and recommended that species with large mature size and greater total leaf area will likely contribute more toward system hydrologic function.

While plants have a demonstrated impact on the hydrologic and pollutant removal performance of bioretention practices, relatively few studies have investigated the specific role of trees in these systems. Instead, the majority of research on vegetation in bioretention has focused on hardy species of grasses and shrubs, which can tolerate the dynamic soil moisture conditions in bioretention practices. Urban trees provide a number of ecosystem services, such as mitigating the heat island effect (Kurn et al., 1994), removing airborne pollutants and improving air quality (Nowak et al., 2006), and influencing urban hydrology through the processes of interception, stemflow, and throughfall (Xiao and McPherson, 2016). Trees may serve an important role in the ability of bioretention practices to manage stormwater runoff while also impacting urban sustainability by incorporating ancillary environmental and social benefits (e.g., Mason et al., 2017). Further, a recent study by Tirpak et al. (2018) showed that while some tree species are not tolerant of the harsh bioretention environment, some trees appear to be well adapted to these conditions. However, a better understanding of the function and performance of trees in bioretention practices is needed to fully recognize their potential role in urban stormwater management.

To address this need, a mesocosm-scale study was conducted to examine the hydrologic and pollutant removal contributions of various tree species in bioretention practices. Semi-synthetic stormwater was applied to bioretention columns containing various tree species in a controlled environment to investigate differences in pollutant removal performance over a period of 14 weeks. Data-logging scales were utilized to identify differences in hydrologic impacts between tree species compared to nonvegetated mesocosms. The objective of this study was to investigate the role of trees in bioretention practices and identify characteristics related to performance variability. Findings from this research provide insights to urban foresters, stormwater engineers, and regulatory agencies on how to integrate trees into bioretention practices and quantify their contributions to urban stormwater management.

4.3 Materials and Methods

Twenty bioretention mesocosms were installed in a climate-controlled greenhouse in Knoxville, Tennessee, USA in the fall of 2016, where mean daily and nightly temperatures were 29.5°C and 24.5°C over the course of the study, respectively. The mesocosms were constructed using 208L repurposed high-density polyethylene barrels, each with a diameter of 610mm and height of approximately 1050mm. This diameter is unique in literature, being larger than those utilized in most studies. This was intentional, as the mesocosms provided additional space for tree growth during the study, an important consideration to avoid root restriction issues and unrealistic root:soil volume ratios associated with overly small planting containers. Each mesocosm contained approximately 760mm of bioretention media, consisting of 93% sand, 7% clay, and 5% organic matter (by weight) in the form of pine bark mulch, topped with a 75mm layer of shredded hardwood mulch and underlain by a layer of small diameter gravel and washed stone to both prevent media washout and facilitate drainage through the port at the bottom of the column. Twelve mesocosms were placed on data-logging drum scales, which logged measurements every minute at a resolution of 45g, to observe changes in weight due to drainage and ET after watering. A diagram of the components of the mesocosms is shown in Figure 4.1.

Four mesocosm planting treatments were utilized in this study: three native US tree species (*Acer rubrum* – red maple, *Pinus taeda* – loblolly pine, and *Quercus palustris* – pin oak) and a nonvegetated configuration used as a control throughout the experiment. Each treatment was replicated five times, with three replicates of each placed on scales. Tree species selections were based on commonly used urban trees in the southeastern United States, recommended vegetation in bioretention literature, and the tolerance of the species to the wide range of soil moisture conditions typically found in bioretention practices. Five bare-root, two-year old seedlings of each tree species were randomly planted in the bioretention mesocosms and given approximately seven months to allow for plant establishment. During this establishment period, the mesocosms were watered with tap water on a weekly basis.

Semi-synthetic stormwater applications were conducted over a period of 14 weeks between June and October 2017. The rate and volume of applications were based on 30 years of historic rainfall data for Knoxville, Tennessee, USA (mean of 80 storm events per year and median rain event of 5mm – historic rainfall data not presented), and a simulated drainage area to

Figure 4.1: Cross-section of bioretention mesocosm components.

treatment area ratio of 15:1. Based on these parameters, each watering session consisted of distributing approximately 18L of semi-synthetic stormwater solution to each mesocosm at a frequency of five applications every three weeks (i.e., one week where only one application was performed followed by two consecutive weeks containing two watering events each week). Pollutant levels typically found in worldwide urban runoff were used as target influent concentrations, as presented by Bratieres et al. (2008). Sediment was collected from a local stormwater detention basin, dried, passed through a 300μm sieve, and added as a source of total suspended solids (TSS) in the semi-synthetic stormwater mixture, also following methodology from Bratieres et al. (2008). After determining the contributions from pollutants adsorbed to the sediment as well as baseline levels found in the tap water used to create the semi-synthetic stormwater solution, various chemicals were added to achieve the desired influent concentrations (Table 4.1). The semi-synthetic stormwater mixture was continuously mixed in a 750L tank during watering sessions and was distributed in three phases to the mesocosms to ensure uniform dispersion of constituents were maintained in the stormwater applied to each column. It should be noted that the nitrogen species in the semi-synthetic stormwater mixture were present at higher concentrations than typical stormwater levels and values reported in previous mesocosm studies (e.g., Read et al., 2008, Bratieres et al., 2008). This occurred even after reducing the dosing rates of chemical sources of nitrogen amendments to the stormwater mixture, indicating these elevated concentrations were likely attributed to nitrogen pollutants adsorbed to the sediment collected from the stormwater detention basin.

Weekly samples were collected to monitor the water quality performance of the mesocosms. Inflow samples were composited by collecting samples directly from the outlet of

Table 4.1: Mean pollutant concentrations in semi-synthetic stormwater applications during testing and chemical sources used in mixture. Coefficients of variation (in %) for each constituent are listed in parentheses.

Pollutant	Concentration	Method Detection Limit (MDL)	Source
TSS (mg L^{-1})	75 (26.7)	0	Stormwater sediment
NH_4^+-N (mg L^{-1})	0.39 (135.7)	0.02	NH_4CL
NO_x-N (mg L^{-1})	3.62 (4.0)	0.01	KNO_3, other N sources
PO_4^{3-} (mg L^{-1})	0.17 (85.1)	0.12	KH_2PO_4
Cu (µg L^{-1})	67 (24.1)	1	Standard Cu solution
Pb (µg L^{-1})	51 (46.1)	3	$PbNO_3$
Zn (µg L^{-1})	206 (16.0)	16	Standard Zn solution
Cr (µg L^{-1})	18 (30.8)	4	Standard Cr solution
Mn (µg L^{-1})	201 (3.8)	1	Standard Mn solution
Fe (µg L^{-1})	654 (30.9)	4	$FeSO_4$
Ni (µg L^{-1})	23 (9.1)	3	Standard Ni solution
Cd (µg L^{-1})	5 (22.9)	2	Standard Cd solution

the mixing tank during watering sessions, while outflow samples were taken approximately 24 hours after a watering session from containers placed below each column to collect effluent. Samples were analyzed for TSS using USEPA Method 160.2 (USEPA, 2015). After passing samples through 0.45μm filters, ion chromatography (IC) was used to determine levels of ammonium (NH_4^+-N), nitrite and nitrate (combined as NO_x-N), and phosphate (PO_4^{3-}), while inductively coupled plasma atomic emission spectroscopy (ICP-AES) was used for metals analysis (copper (Cu), zinc (Zn), lead (Pb), chromium (Cr), manganese (Mn), iron (Fe), nickel (Ni), and cadmium (Cd)) in accordance with a combination of standard operating procedures developed by the University of Tennessee (UTK, 2007). When constituents in samples were measured below the method detection limit (MDL), a value of 0.5*MDL was used for statistical analysis (Table 4.1).

Data collected from the scales followed a typical decay curve for soils moving from saturated to field capacity soil moisture conditions (Zotarelli et al., 2009). During this decay, distinguishing between drainage and ET was not possible, as they occurred simultaneously. However, as drainage ceased, the data began to exhibit “step-changes” due to the diurnal patterns in ET processes (Figure 4.2). Weight losses associated with ET were identified through these step-changes. During daytime hours, small weight drops occurred as water was removed from the mesocosms as water vapor, either through evaporation (which increased during the day due to rising temperatures) or transpiration (which increased during the day along with increases in photosynthesis). These weight drops then reached relative plateaus during nighttime hours, as evaporation and transpiration lessened. Because these daily step-changes in mesocosm weight were most readily identified in the data beyond 24 hours after a semi-synthetic stormwater application (i.e., after drainage verifiably ceased), ET assessments were conducted during a week-long dry period after a watering event (on weeks when only one stormwater application was conducted). A total of six such instances (i.e., a week-long dry period following a watering session) occurred during the 14-week study; thus, six measurements of ET for each of the twelve mesocosms on scales were conducted.

To quantify the weight changes due to ET and aid in the identification of the step-change behavior in the data, the scale readings were smoothed to reduce noise that was present in the raw data. This noise was likely associated with the use of climate control equipment in the greenhouse, which may have impacted the scale readings due to high power usage. A fifth order lowpass Butterworth filter was applied to the scale data using Matlab, beginning at 24 hours after the watering event to the end of the week-long dry period prior to the next watering session (The MathWorks, 2016). The data were filtered twice (i.e., in the forward and backward direction) to eliminate time lag between the raw and filtered data. Because weight losses due to ET could not be separated from drainage losses during daytime hours, the time when ET became the dominant weight loss process (and weight losses due to drainage were no longer occurring) was determined when hourly weight changes over a six-hour nighttime period (beginning between 21:00 and 6:00) were less than 90g (corresponding to twice the resolution of the scales). Once this ET “start point” was determined, the rate of ET (in mm d^{-1}) was calculated from the weight losses that occurred from the weight at this point to the weight recorded at the end of the week-long dry period. An example of this analysis is shown in Figure 4.2. Transpiration rates of treed columns were then determined by comparing these ET rates to those observed in the nonvegetated columns, which corresponded solely to evaporation rates.

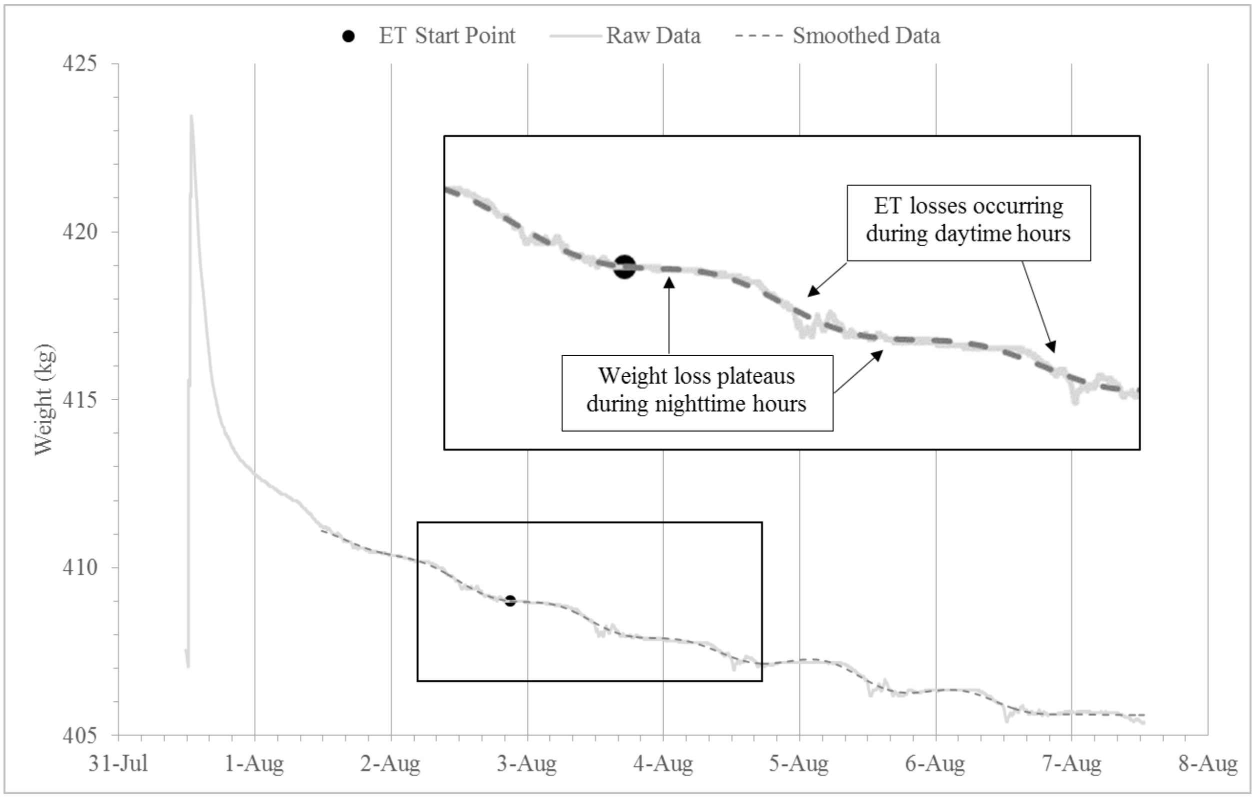

Figure 4.2: Determination of weight losses attributed to evapotranspiration (ET) from a mesocosm planted with a pin oak tree using smoothed scale data following a watering event on July 31, 2017. The start point of losses solely due to ET occurred at 9:00pm on August 2, 2017. Total ET losses from this event were 3.63kg, corresponding to a rate of approximately 2.7mm d^{-1}. The daily step-changes in weight, as well as the raw scale data (solid line), smoothed scale data (dashed line), and ET start point, are magnified for clarity in the inset.

Effluent concentrations and ET rates were compared across the four mesocosm configurations. Results from Shapiro-Wilk tests for normality indicated that the effluent concentrations were not all normally distributed. ET rates were normally distributed after removing outliers from the data, which were identified as values greater than 1.5*IQR beyond the upper and lower quantiles and verified based on the drainage behavior of the given mesocosms. No outliers were identified in the water quality data. Comparisons of pollutant removal performance were made using Wilcoxon Signed Rank tests, and paired t-tests were used to assess differences in mean ET rates (mm d^{-1}) between the mesocosm configurations. Statistically significant differences were considered at $p<0.05$ and $p<0.1$ (when noted), and analyses were performed using the statistical software packages JMP Pro 13.2 (JMP, 1989-2007) and R (R Core Team, 2016).

4.4 Results and Discussion

4.4.1 Effect of Tree Species on Effluent Pollutant Concentrations

Tree species differences had varying degrees of influence on pollutant removal during the study (Table 4.2). No significant differences in effluent TSS concentrations were observed between red maple, loblolly pine, pin oak, and nonvegetated mesocosms. These findings are somewhat expected, due to the established success of bioretention media in TSS removal. It should be noted that all mesocosms would have exceeded minimum levels established in the National Pollutant Discharge Elimination System (NPDES) general permit for stormwater discharges applicable to the project location (Knoxville, Tennessee, USA), which requires new development sites to attain an 80% reduction in TSS from stormwater runoff (TDEC, 2016).

Table 4.2: Mean effluent pollutant concentrations (±SE) for all mesocosm configurations during semi-synthetic stormwater testing.

Pollutant	Effluent Concentration			
	Nonvegetated	Red Maple	Loblolly Pine	Pin Oak
TSS (mg L^{-1})	3±1	5±1	3±1	2±1
NH_4^+-N (mg L^{-1})	0.01±0.00	0.01±0.00	0.01±0.00	0.01±0.00
NO_x-N (mg L^{-1})	0.13±0.03	0.12±0.02	0.17±0.03	0.14±0.03
PO_4^{3-} (mg L^{-1})	0.06±0	0.06±0	0.06±0	0.06±0
Cu (µg L^{-1})	3±0	4±1	3±0	3±0
Pb (µg L^{-1})	4±1	4±1	10±3	4±1
Zn (µg L^{-1})	42±10	36±8	35±7	40±7
Cr (µg L^{-1})	3±0	3±0	4±0	4±0
Mn (µg L^{-1})	339±26$^{\boldsymbol{A}}$	254±26$^{\boldsymbol{B}}$	184±29$^{\boldsymbol{B*}}$	254±18^{B*}
Fe (µg L^{-1})	61±15	103±32	114±28	100±27
Ni (µg L^{-1})	2±0$^{\boldsymbol{A}}$	2±0$^{\boldsymbol{A}}$	8±2$^{\boldsymbol{B}}$	2±0$^{\boldsymbol{A}}$
Cd (µg L^{-1})	2±0	2±0	2±0	2±0

Note: Significant differences (at p<0.05) between mesocosms types determined from Wilcoxon Signed Rank tests are indicated by different letters. Significant differences (p<0.05) between pin oak and loblolly pine trees for Mn are indicated by an asterisk (*).

Similarly, no significant differences in effluent NH_4^+-N, NO_x-N and PO_4^{3-} concentrations were observed between any of the mesocosm configurations. These results suggest that uptake via tree roots was not a significant removal pathway of nutrients from the mesocosms. This is contradictory to findings from previous studies, especially for nitrogen compounds, which reported that plants enhanced nitrogen removal in bioretention practices through root uptake (i.e., Lucas and Greenway, 2008, Read et al., 2008). One possible explanation for this finding is that nitrogen removal via soil/microbial processes played a greater role than plant uptake due to the larger mesocosms used in this study (208L). Read et al. (2008) used mesocosms with a volume of roughly 9L, which may have allowed plant roots to occupy more of the available soil matrix and consequently increased nitrogen uptake levels compared to the larger mesocosms used in this study. Lucas and Greenway (2008) used similarly large (240L) mesocosms in their study and found that, while vegetated mesocosms removed more total nitrogen (TN) than nonvegetated columns, TN removal levels exceeded the rate of nitrogen uptake by plants used in the study, suggesting denitrification was contributing to nitrogen removal. Among other influencing factors, such as soil properties and species tolerances to nutrient levels, plant uptake of nutrients in bioretention practices may vary depending on root structure and distribution within the bioretention media profile, and long-term monitoring may be needed to determine uptake rates. In a natural system, total N uptake by trees has been estimated at 32 to 114 kg ha^{-1}, but in the short term, soil nitrogen pools were found to be highly variable and driven primarily by microbial processes (Nadelhoffer et al., 1984). Nitrogen and phosphorous uptake occurs when roots (or root-associated mycorrhizae) directly intercept nutrient deposits in soils, as well as via ion movement and water flow through soils along gradients toward roots (Pallardy, 2008). As the trees mature and root systems become more established and occupy a greater volume of the bioretention media, plant uptake would be expected to serve a greater role in nitrogen removal from the mesocosms.

Significant differences in effluent concentrations from the mesocosms were observed for some metals. Loblolly pines had significantly higher effluent Ni concentrations than red maple, pin oak, and nonvegetated mesocosms ($p<0.05$), though no other significant differences for Ni removal were observed. Pin oaks had higher effluent Mn concentrations than loblolly pines ($p<0.05$), though this was the only observed difference in Mn removal between tree species. Mesocosms planted with trees had significantly lower Mn effluent concentrations than nonvegetated mesocosms ($p<0.05$), though both resulted in a net production of Mn. This is consistent with findings from Read et al. (2008), who also reported elevated Mn and Fe effluent concentrations and attributed them to reduced oxygen levels deeper in the media profile which resulted in Mn and (to a degree) Fe precipitation. Mn is an essential micronutrient for trees and is involved in chlorophyll synthesis and photosynthesis, which may explain why treed mesocosms had lower effluent concentrations than the nonvegetated mesocosms (Pallardy, 2008). No significant differences were observed between any configurations for effluent concentrations of Cd, Cr, Cu, Fe, Pb, and Zn, consistent with previous studies which have linked metals removal to complexation sites in bioretention media, which was present in all mesocosms (Hunt et al., 2012, Wang et al., 2017). Finally, though metal removal performance varied across the mesocosms to a degree, effluent concentrations of metals commonly analyzed in urban stormwater runoff (i.e., Cu, Pb, and Zn) were not significantly different between the four configurations in the study, indicating the role of the bioretention media in the removal of these species.

Though few have specifically investigated the contribution of trees, the role of vegetation in heavy metals removal in bioretention practices has been investigated in previous studies with varied results. Muthanna et al. (2007) studied water quality performance in a pilot-scale bioretention box and found that between 2% and 7% of heavy metal removal could be attributed to the shrubs and flowering species planted in the systems via assimilation into roots and leaves. Feng et al. (2012) found that plants in bioretention columns planted with shrubs, grasses, sedges, and perennials significantly influenced the removal of Fe, Cr, and Al, however Cu, Pb, and Zn levels were unaffected by the presence of vegetation. As was the case in this study, Read et al. (2008) found some variation in metals removal (i.e., Mn and Zn) between a small subset of the plant species studied, though effluent metal concentrations from planted trials did not differ from soil-only controls, similar to findings reported in Hatt et al. (2007). Many metal species are used by trees as micronutrients and play key roles in metabolic and physiological processes (Pallardy, 2008). However, because they are only needed by trees in small concentrations, the metals removal provided by the bioretention media may dampen any differences between species. As the trees grow, it would be expected that the uptake of metals for physiological processes would have a larger effect on the removal of metals in bioretention practices, though species differences in contributions to metal removal performance may not be evident until trees have reached a sufficiently mature size.

Overall, though some differences in pollutant removal were present between the configurations, comparing the results of the mesocosms showed largely consistent performance in removing constituents from the influent semi-synthetic stormwater. Aside from Mn (which was exported from the mesocosms), effluent concentrations from all mesocosms for all pollutants were significantly lower than influent concentrations ($p<0.01$). These results suggest that the bioretention media, the only component common to all configurations, was critical to pollutant removal via soil-based processes over the course of the study. This finding illustrates the role of the media in removing dissolved and particle-bound constituents from influent runoff, and the importance of media composition specifications and testing prior to installation. It also highlights the need to understand how much root volume is truly present in field-scale bioretention practices and how it changes over time, as the role of plants in system performance may be dependent on this attribute. Such information will help scale results from mesocosm-scale studies to full-scale field installations.

4.4.2 The Role of Evapotranspiration in Mesocosm Hydrology

Results from the assessment of ET rates in the mesocosms are presented in Table 4.3. Differences in mean ET rates for all treed configurations compared to the nonvegetated mesocosms were significant at $p<0.05$ aside from pin oak ($p<0.1$). Mesocosms planted with red maple trees exhibited significantly higher ET rates than all other configurations ($p<0.05$). These results may be connected to the continuous rapid growth of the red maple trees used in the study, which resulted in more numerous leaves and a visibly greater total leaf area than the other species by the end of the study. However, as with other deciduous species, ET rates in the mesocosms planted with red maple trees would be expected to decline in the fall and winter months as the trees shed their leaves and enter dormancy. Mean ET rates for mesocosms planted with pin oaks and loblolly pines were not significantly different. However, over time, it would be expected that differences in ET between the pin oak and loblolly pine trees would become evident due to differences in physiology, seasonal impacts on tree function, tree growth, and increased canopy size compared to the seedlings used in this study.

The ET rates observed in the mesocosms are somewhat comparable to values reported in other studies investigating the role of ET in bioretention hydrology. The mean ET rate observed in the mesocosms planted with red maple (3.2 mm d^{-1}) was similar to the mean rate of 3.1 mm d^{-1} from a freely draining mesocosm planted with native grasses studied by Wadzuk et al. (2015), though all other configurations fell below this rate. Mean ET rates of all mesocosms were also well below the 6.1 mm d^{-1} ET rate reported by Wadzuk et al. (2015) from a bioretention mesocosm constructed with an internal water storage (IWS) layer, and from ET values observed by Denich and Bradford (2010), who reported average ET rates of 4.2 mm d^{-1} in a bioretention practice during sunny, dry weather using lysimeters. Average ET rates from rain garden mesocosms reported by Hess et al. (2017) were between 2.7 mm d^{-1} to 4.3 mm d^{-1} depending on media type and drainage configuration, comparable to the rates observed in this study.

Given that the trees used in this study were planted as seedlings, the similarities between previously reported ET rates in bioretention practices and the rates observed in the mesocosms planted with trees are promising. If the trees continued to mature, or had larger trees been planted in the mesocosms at the commencement of the study (as would likely be the case in field-scale installations), ET rates and water uptake would be expected to increase due to additional transpirational leaf area and rooting volume. While seedlings the size of those used in this study typically have a leaf area of less than 1 m^2, a ten-year-old tree 20cm in diameter has a leaf area of 100 m^2 and thus the potential for a 100-fold increase in ET as the tree becomes established (Peper et al., 2001). However, further research on ET rates of full-scale bioretention practices

Table 4.3: Mean evapotranspiration (ET) rates observed during the study as determined from data-logging scale data. Values are presented as mean ET rate ±SE.

Mesocosm Configuration	ET Rate (mm d^{-1})
Nonvegetated	2.01±0.10
Loblolly Pine	2.21±0.12
Pin Oak	2.19±0.08
Red Maple	3.22±0.20

planted with trees is needed to confirm this hypothesis. Finally, while reporting losses attributed to ET represents the total amount of water exiting the system as water vapor, isolating the transpiration component from ET may provide a more useful insight to stormwater engineers selecting tree species for future bioretention practices.

4.4.3 The Role of Transpiration in Mesocosm Hydrology

The significant differences between treed and nonvegetated mesocosms highlights the contributions of transpiration to water losses from the systems compared to evaporation alone. Comparing the mean ET rates observed in the treed mesocosms to the nonvegetated mesocosms, transpiration rates ranged from a minimum of 0.18 mm d^{-1} for the pin oaks to a maximum of 1.21 mm d^{-1} for the red maples, accounting for approximately 8.2-37.5% of average daily ET losses. Transpiration is influenced by many plant-specific factors, including leaf area, size, shape, orientation, concentration of stomata on leaf surfaces, degree of stomatal control, root-shoot ratio, and tree age and size (Pallardy, 2008). Conditions associated with the local microclimate such as temperature, precipitation, vapor pressure deficit, and soil moisture also influence water losses via transpiration. Further, trees grown in bioretention practices may be exposed to several factors specific to the urban environment that may influence growth and function, including compacted, degraded soils, exposure to anthropogenic contaminants, limited nutrient and water availability, etc., and lead to potential deviations in transpiration behavior from natural settings (Day et al., 2010, Craul, 1985). Therefore, comparisons between transpiration rates measured in this study to transpiration values reported in forestry literature for these tree species may not adequately account for these differences in growing condition.

Physiological differences, such as xylem anatomy (i.e., ring- versus diffuse-porous), between the tree species used in this study may influence the rate of water movement through the stem, though they may not best explain transpiration differences at the whole-tree scale over longer time periods (Pallardy, 2008). Instead, based on findings from Guidi et al. (2008), who found that ET rates in willow and poplar trees planted in vegetated filters were more strongly tied to plant development and nutrition rather than differences between species, the differences in tree-level ET rates observed between the three species may be better explained by the health and growth rate of the trees during the study. Though plant nutrition was not directly monitored through foliar nutrient testing, plant development can be tied to canopy size and stem diameter. During the study, the red maples produced much larger, more densely vegetated crowns than the loblolly pines and pin oaks. Leaf area index and dimensions of the crowns were not directly measured to quantify differences in vegetation. However, because of the allometric relationship between canopy size and diameter (i.e., increasing canopy size requires similar increases in cross-sectional area connected to the development of conducting xylem tissue to meet the water demands associated with increased vegetation), comparisons between the diameter of the stems can be made in place of canopy size and dimension (Pallardy, 2008, Pretzsch et al., 2015).

Tree diameters were measured approximately 10cm above the root collar one week prior to commencing semi-synthetic stormwater applications. The average diameters for the trees grown in mesocosms installed on the data-logging scales were 21mm, 17mm, and 15mm for the red maple, loblolly pine, and pin oak trees, respectively. Due to the scope of the study, there are not enough replications to assess the level of statistical significance between these diameter measurements. However, they are in-line with observed differences in canopy size between the species and support the trends in measured ET rates in Table 4.3. Because the trees were of similar size and age when planted in the mesocosms, the diameter measurements, in conjunction with relative canopy size and ET rates, suggest that red maples experienced a greater degree of development and growth relative to the other species during the plant establishment period and throughout the study, and highlight the importance of selecting tree species that can tolerate (and succeed in) the unique growing conditions found in bioretention practices (i.e., prolonged periods of drought followed by intermittent inundation, nutrient deficient, sandy soils, exposure to pollutants present at potentially elevated levels present in urban runoff, etc.). To expand the current understanding of the role that trees may play in bioretention practices, further research is needed to analyze the suitability of additional tree species to the bioretention environment and to investigate how physiological differences influence tree performance on a seasonal basis.

4.5 Conclusions

The hydrologic and pollutant removal performance contributions of trees in bioretention practices were studied by dosing twenty bioretention mesocosms of four vegetated treatments, including three native US tree species (*Acer rubrum* – red maple, *Pinus taeda* – loblolly pine, and *Quercus palustris* – pin oak) and a nonvegetated control, with a semi-synthetic stormwater solution over a period of 14 weeks. Major conclusions from this research include the following:

- Comparing the water chemistry of influent stormwater to effluent samples collected from each mesocosm revealed primarily nonsignificant differences between the configurations, likely attributable to the relatively small volume of media occupied by the roots of the seedlings used in this study relative to other studies.

- Daily evapotranspiration (ET) rates, characterized using data-logging scales, were significantly higher in mesocosms planted with trees compared to nonvegetated mesocosms, demonstrating the role of transpiration in bioretention hydrology, which accounted for between 8.2-37.5% of average daily water losses from the mesocosms.
- The average ET rate from mesocosms planted with red maple trees (*Acer rubrum*) (3.2 mm d^{-1}) was significantly larger than all other configurations, potentially due to the degree of plant development, canopy size, and growth compared to the other species used in the study.

As with other plants, tree species suitability for the bioretention environment should be considered when incorporating trees into bioretention planting plans. Through careful assessment of tree suitability and species selection, stormwater engineers may improve the hydrologic and pollutant removal performance of bioretention practices through the inclusion of trees, while increasing the overall environmental impact of these systems by incorporating the various ecosystem services that urban trees provide. Future research should investigate the performance of larger, more mature trees in bioretention practices, whose roots would occupy a greater soil volume, potentially increasing the role of plant uptake in pollutant removal performance in these systems. Studies should also examine the impact physiological differences between species have on seasonal transpiration levels and how these variations may influence bioretention performance. Finally, studies should expand upon the number of species used in the study and alter the composition and dosing frequency of the semi-synthetic stormwater solution to analyze tree suitability and performance for bioretention practices outside of the southeastern United States.

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CHAPTER 5 : HYDROLOGIC AND POLLUTANT REMOVAL PERFORMANCE OF SUSPENDED PAVEMENT SYSTEMS FOR URBAN STORMWATER MANAGEMENT

5.1 Abstract

Trees supply numerous ecosystem services to the urban environment, including mitigating the urban heat island, improving air quality, and providing habitat for wildlife. However, due to the structural stability requirements of infrastructure such as sidewalks, roadways, and parking areas, urban soils are commonly characterized by high compaction levels, low porosity, and nutrient deficiency, often to the detriment of urban tree health. By transmitting surface loads to a compacted subbase, suspended pavement systems create a matrix of uncompacted soil that promotes tree health through increased root access to oxygen, water, and nutrients. When backfilled with an engineered bioretention media, suspended pavement systems can also provide an opportunity for subsurface stormwater management in ultra-urban areas where space may be limited due to concentrated development and high land costs. Two suspended pavement systems designed to function as subsurface bioretention practices were installed in Knoxville, Tennessee, USA, in 2015 and monitored over 27-months. During that period, over 99% of runoff volumes were reduced by the free-drained north system, which completely captured runoff from 79% of storms. The underdrained south system reduced influent runoff by over 88% and captured all runoff from 83% of events during the study. Influent TSS concentrations were significantly reduced by the south suspended pavement system, though no other significant differences between influent and effluent pollutant concentrations were observed, presumably due to low influent concentrations. This study demonstrates the viability of suspended pavement systems in a stormwater management application and illustrates the hydrologic and pollutant removal capabilities of these systems to manage urban stormwater runoff.

5.2 Introduction

The presence of trees in urban areas has been shown to provide multiple ecosystem services to urban populations. Trees mitigate the urban heat island and reduce energy consumption through shading and evaporative cooling (Akbari, 2002, Livesley et al., 2016) and improve air quality through the interception of particulate matter on vegetated surfaces and adsorption of gaseous air pollutants (Nowak et al, 2006, McPherson, 2016). Trees planted alongside streets provide habitat for wildlife, attenuate vehicle noise, and protect pedestrians from motorists (Mullaney et al., 2015). Trees influence urban hydrology through the processes of interception, stemflow, and throughfall, and by enhancing infiltration rates through macropores created in the soil by roots (Xiao and McPherson, 2016, Asadian and Weiler, 2009, Pataki et al., 2011, Bartens et al., 2008). Trees also impact the hydrology of the urban landscape through transpiration. Recently, a study by Tirpak et al. (2018a) showed that transpiration rates of trees planted in green infrastructure practices can be influenced by design strategies which promote higher soil moisture conditions to enhance their contributions to urban stormwater management.

However, many aspects of the urban environment, especially poor soil conditions, present challenges to tree survival and growth in cities (Day et al., 2010). Because of disturbances from land development and structural stability requirements for roadways, sidewalks, and other infrastructure, urban soils are typically characterized by high levels of compaction, decreased porosity, poor aeration and water availability, and nutrient deficiency (Craul, 1985). Highly compacted urban soils and nearby belowground impermeable infrastructure (e.g., building foundations) create physical barriers that restrict root exploration which, in combination with the lack of available water or nutrients in urban soils, negatively impacts tree growth (Craul, 1985, Day et al., 2010, Krizek and Dubik, 1987). Tree-specific design techniques such as suspended pavement systems have been implemented in recent decades to maintain the soil stability necessary for paved urban infrastructure while addressing the detrimental impacts of compacted urban soils on tree health (Bartens et al., 2010).

Suspended pavement systems are designed to transmit the load from vehicles, pavement, and pedestrians to a compacted subbase, creating an uncompacted matrix of soil that provides a more accommodating environment for tree roots (Page et al., 2015). Suspended pavement systems can be comprised of concrete pillars or columns installed below paved surfaces, as well as commercially-available, propriety devices such as the Silva Cell (Deeproot) or RootSpace (GreenBlue Urban) (Page et al., 2015). Studies have found that these practices improve tree growth/health and provide greater stability compared to other trees grown in compact, nutrient deficient urban soils (Smiley et al., 2006, Bartens et al., 2010). Research has also shown that the uncompacted soil matrix created by suspended pavement systems can be designed to function as a bioretention practice, a type of stormwater control measure (SCM) commonly implemented in cities to manage urban stormwater runoff (Page et al., 2015).

Bioretention practices utilize soil and plant processes to mitigate the quantity and improve the quality of runoff produced in urban areas (Davis et al., 2009). Stormwater runoff that is routed into a bioretention practice infiltrates through the sandy engineered media, where pollutants are filtered out, adsorbed to soil particles, or taken up by plants before the treated water is released into underlying subsoils or transported via connections to an existing sewer network (Davis et al., 2009). Research has shown the ability of bioretention practices to lessen the impacts of urban stormwater to receiving waterways through volume reduction and peak flow mitigation (Hunt et al., 2006, Li et al., 2009, Olszewski and Davis, 2013), as well as removal of pollutants commonly found in runoff, such as total suspended solids (TSS), nutrients (i.e., nitrogen (N) and phosphorous (P) species), and heavy metals (e.g., copper (Cu), lead (Pb), and zinc (Zn)) (Passeport et al., 2009, Chapman and Horner, 2010, Brown and Hunt, 2011).

By utilizing bioretention media and routing stormwater runoff into the matrix of uncompacted soil, suspended pavement systems can provide a subsurface green infrastructure alternative while promoting tree health and increasing the ancillary environmental benefits provided by trees in urban areas. This is especially advantageous in ultra-urban areas, where space for conventional green infrastructure may be limited due to concentrated development and/or high land costs. In a study that investigated this application of suspended pavement systems, Page et al. (2015) found that peak flow rates were reduced by 62% and significant removal was observed for several stormwater pollutants by two suspended pavement systems installed in Wilmington, North Carolina, USA. While the systems studied by Page et al. (2015) were shown to effectively manage stormwater runoff, the designs used in the study (i.e., systems lined with an impermeable membrane) are not common outside of research settings, and the interaction between the suspended pavement systems and the surrounding soils was not investigated.

To add to the limited research of suspended pavement systems designed to function as SCMs, the hydrologic and pollutant removal performance of two suspended pavement systems were monitored over a 27-month period in Knoxville, Tennessee, USA. The objective of this study was to provide another characterization of the ability of suspended pavement systems to contribute to urban stormwater management and to determine the influence of design parameters and drainage configuration on system performance.

5.3 Materials and Methods

5.3.1 Site Description

The study was located in Knoxville, Tennessee, USA (35.9606°N, 83.9207°W, approximate elevation of 270m) on the campus of the University of Tennessee. Weather conditions in Knoxville are consistent with a temperate climate, with a mean annual temperature of 16°C and mean annual precipitation of 1215mm (Tennessee Climatological Service). Two suspended pavement systems (hereafter referred to as the "north site" and "south site" due to their geographic orientation) were installed in the winter of 2016 to manage stormwater produced from two completely impervious, directly connected drainage areas located approximately 100m apart, each comprised of a small segment of a two-lane road (Figure 5.1). The drainage areas of the north and south sites were 183.0m^2 and 138.5m^2, respectively. The cells were designed based on a static storage approach at the system surface (see NCDEQ, 2017), whereby the entire water quality volume could be stored within a 10cm ponding zone at the top of the system. The subsequent required surface area of the practices (along with the two-tiered configuration) also approximated the soil volumes recommended by the manufacturer (Deeproot) for mature trees of roughly 20-25cm diameter at breast height (DBH) (16-20 m^3), providing for tree growth in the practices over time. Both systems were of similar size; the surface area of the north site (22.3 m^2) represented 12% of the contributing drainage area, while the surface area of the south site (27.0 m^2) corresponded to 19% of its drainage area. These large loading ratios were primarily the byproduct of the static design approach and the limited available storage volumes present in these systems. Design components of the two suspended pavement systems are outlined in Table 5.1.

During construction, excavated pits were lined with a 10cm gravel subbase upon which the Silva Cell frames were installed. At the south site, three perforated 10cm diameter polyvinyl chloride (PVC) pipes served as an underdrain system to convey the portion of runoff that did not exfiltrate into underlying soils to a monitored outfall. At the north site, water that had percolated through the bioretention media drained exclusively to underlying subsoils (or exited via overflow). The practices were backfilled with bioretention media in a series of lifts to a depth of 10cm below the Silva Cell decks. Influent runoff was routed into the practices via curb cutouts and was distributed across the top surface of the media in each system through a network of three perforated 10cm diameter corrugated drainage pipes placed directly below the Silva Cell decks. Overflow pipes (10cm diameter corrugated drainage pipes) were installed to bypass the suspended pavement systems during large rain events and route runoff away from the practice (south site) or into an existing catch basin (north site). Overflow was possible if the influent flowrate exceeded the distribution system capacity, if inflow exceeded surface infiltration capacity, or if the system was completely saturated. After installation, the decks were covered with a geotextile fabric and topped with topsoil and turf grass, which was used in place of pavement due to local constraints. Based on species selection recommendations reported in Tirpak et al. (2018b), one bald cypress (*Taxodium distichum*) tree of approximately 5cm DBH was planted in each system in March 2017, and root barrier devices were installed to direct root growth into the bioretention media (Figure 5.2).

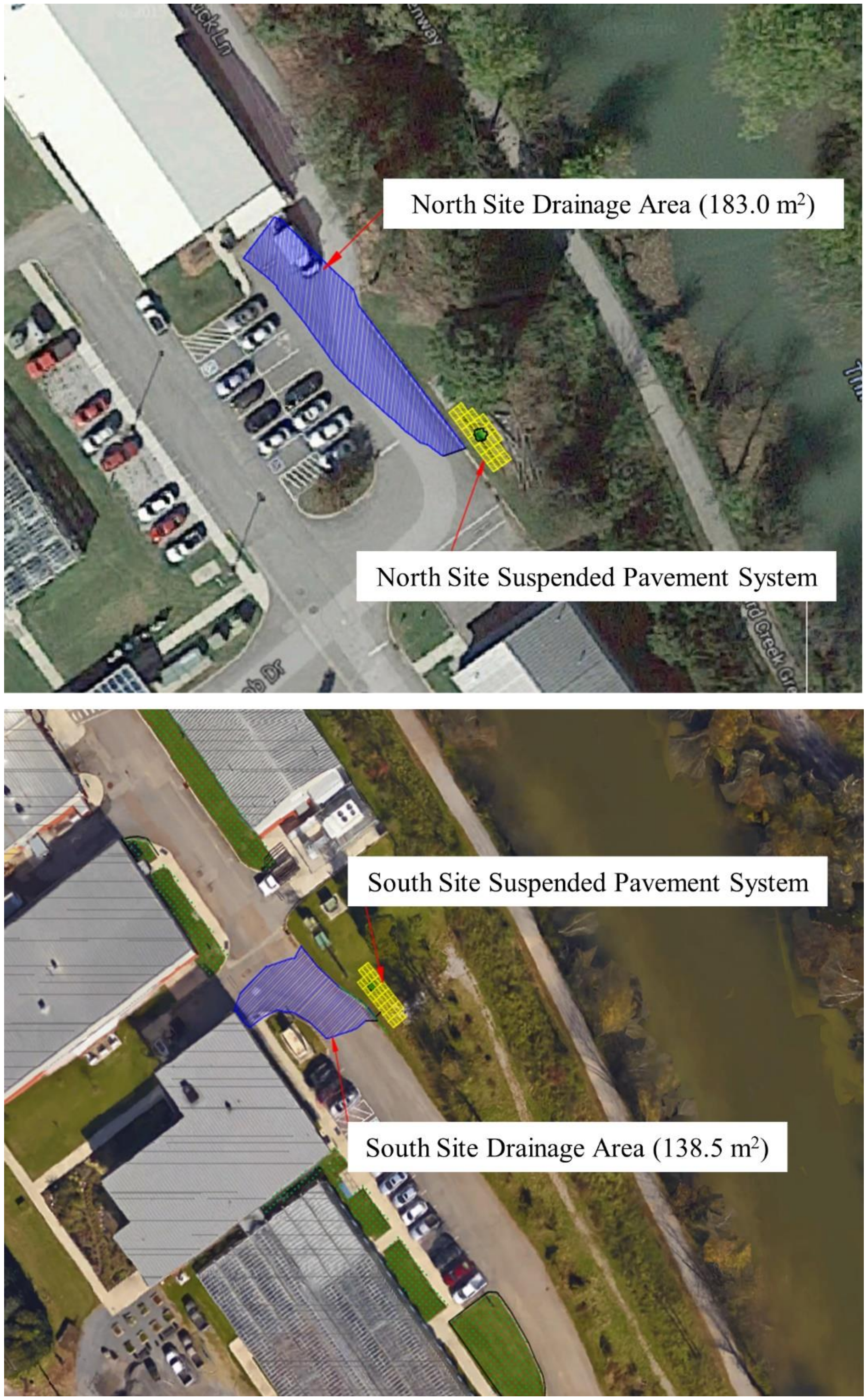

Figure 5.1: Location of north (top) and south (bottom) suspended pavement systems and contributing drainage areas.

Table 5.1: Summary of suspended pavement system design components and drainage area characteristics.

Parameter	North Site	South Site
Drainage area (m^2)	183.0	138.5
Imperviousness (%)	100	100
Design storm event (mm)	25.4	25.4
Treatment surface area (m^2)	22.3	27.0
Approx. loading ratio	8:1	5:1
Silva Cell Decks	28	35
Silva Cell Frames	56	70
Media volume (m^3)	15.9	19.2
Bioretention media depth (cm)	71.1	71.1
Media composition	93% sand, 7% fines	93% sand, 7% fines
Organic matter (by weight) and source	5% pine bark mulch	5% pine bark mulch
Gravel subbase thickness (cm)	10	10
Average available ponding depth (cm)	10	10
Estimated infiltration rate (cm hr^{-1})[1]	0.08	0.10
Drainage configuration	No underdrain	Underdrain
Underdrain diameter (cm)	-	10
Vegetation	Bald cypress	Bald cypress

[1]Note: Estimates of infiltration rates of underlying soils based on drawdown rates recorded in perforated well pipes.

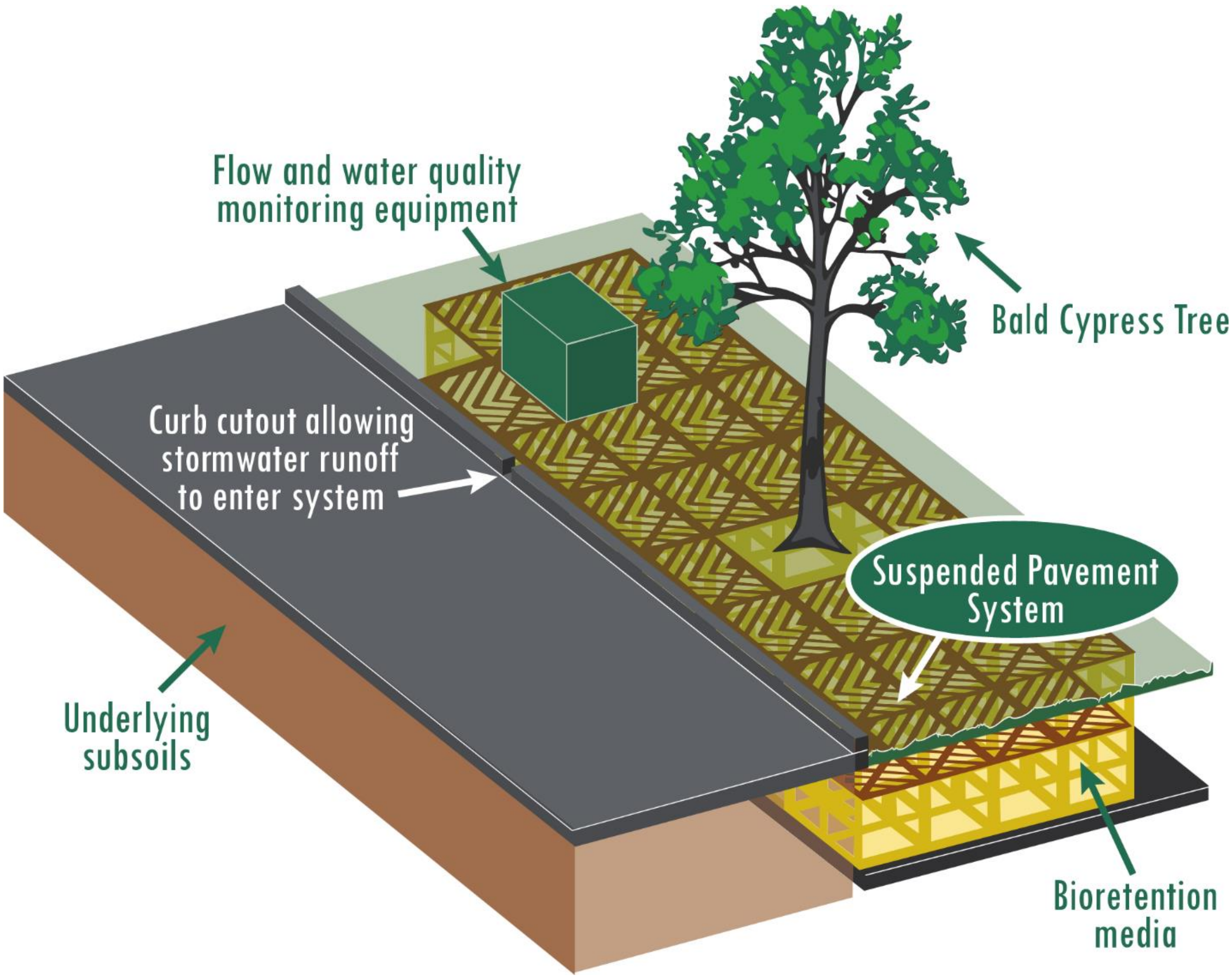

Figure 5.2: Rendering of suspended pavement system components and surrounding infrastructure.

5.3.2 Monitoring Setup

Rainfall measurements were recorded every minute using a 0.254mm ISCO 674 tipping bucket rain gauge (Teledyne Isco) installed in an open area free of overhead obstructions directly adjacent to the south site. ISCO 6712 autosamplers (Teledyne Isco) were installed at the inlet and outlet of the south practice to collect flow-paced, composited water quality samples. Inflow was routed through a 0.12m (0.4ft) fiberglass HS flume (Tracom Inc.) installed at both sites. Outflow from the south site was routed to a 30° sharp-crested, v-notch weir, in which stage was measured using an ISCO 730 bubbler flow module (Teledyne Isco). Overflow from both sites was routed to 45° sharp-crested, v-notch weirs and stage was measured using HOBO U20 water level loggers (Onset Computer Corporation). Stage was converted to flow via standard equations. Similar water level loggers were placed in perforated 10cm PVC vertical well pipes to measure internal water levels relative to the top of the gravel subbase in each practice. All water level loggers recorded data every minute and readings were later corrected to gauge pressure using ambient atmospheric pressure data from an additional logger placed in an on-site monitoring box.

5.3.3 Data Collection and Analysis

Hydrologic data were managed and analyzed using Flowlink version 5.1 (Teledyne Isco), Hoboware (Onset Computer Corporation), and Excel (Microsoft) software packages. Individual storm events were separated by a minimum antecedent dry period of 6hr. Influent runoff volumes entering the systems were determined using the NRCS curve number (CN) method, where a CN of 98 (representative of impervious surfaces) was applied to the entire drainage area for both practices (USDA, 1986). Inflow rates to both sites were initially intended to be measured using the HS-flumes; however, the curve number method was ultimately selected due to inconsistencies in water level measurements and unreliable instrument operation. Peak flow rate reductions were not considered in this study but were likely significant due to the rare occurrences of substantial outflow events. Water quality samples were collected from the field and processed within 24hr following a rainfall event. Total suspended solids (TSS) were analyzed using USEPA Method 160.2 (USEPA, 2015). Samples for other parameters were passed through 0.45μm filters, after which ion chromatography (IC) was used to characterize levels of NH_4^+-N, NO_x-N, and PO_4^{3-}, and analysis of Cu, Pb, and Zn was conducted using inductively coupled plasma atomic emission spectroscopy (ICP-AES) following standard operating procedures developed by the University of Tennessee (UTK, 2007). When levels of pollutants in samples were measured below the method detection limit (MDL), a value of 0.5*MDL was used in analysis (USEPA, 1993).

5.3.4 Statistical Analysis

Statistical analyses were performed using the statistical software packages JMP Pro 13.2 (JMP, 1989-2007) and R (R Core Team, 2016). Tests for normality were conducted using Shapiro-Wilk tests, which indicated that the water quality data points were not normally distributed and required the use of nonparametric statistics. Paired differences in pollutant removal were tested using Wilcoxon Signed Rank tests. Significant differences were considered at $p<0.05$ unless otherwise specified.

5.4 Results and Discussion

5.4.1 Hydrology

The two suspended pavement systems were monitored from April 2016 through July 2018. During this period, 146 storm events representing a total of 1809mm of rainfall were successfully monitored at the north site. Similarly, 148 individual storm events, which amounted to a total rainfall of 1922mm, were recorded at the south site. The magnitude of the monitored rainfall events ranged from 1mm to 73mm and the mean and median events at both sites were approximately 13mm and 8mm, respectively. The design storm (25.4mm) for both suspended pavement systems corresponded to the 88th and 87th percentile rainfall event size observed over the course of monitoring period for the north and south sites, respectively.

Due to the large treatment area of the suspended pavement systems relative to the contributing drainage areas based on a static capture volume design (Table 5.1), the majority of the water balance at both sites was comprised of exfiltration and evapotranspiration (ET), and relatively few instances of drainage (outflow) or bypass (overflow) were observed (Table 5.2). Approximately 99.8% and 88.7% of all runoff volumes were exfiltrated into surrounding soils or released into the atmosphere as ET from the north and south sites, respectively. These results are interesting given the low estimated infiltration rates of the underlying soils (Table 5.1). However, water level data collected form the perforated well pipes in each practice indicate that water was quickly draining from the upper layers of the practice, likely due to the local infrastructure (i.e., the two-lane road) adjacent to the practices. Of the 146 events that occurred at the north site, 30 events resulted in overflow. Similarly, 25 of the 148 events monitored at the south site resulted in outflow and/or overflow. Approximately 79% of storms were completely captured by the north system, with zero overflows produced for storms ranging from 1mm to 73mm. Similarly, 83% of all events were completely captured by the south suspended pavement system, which produced zero outflow/overflow for storms ranging from 1mm to 37mm.

Figures 5.3 and 5.4 illustrate the effect of rainfall size (mm) and duration (hr) on the occurrence of outflow/overflow for the north and south sites, respectively. As presented by Li et al. (2009), the slope of the regression lines of the “no flow” events correspond to the intensity (slope) of an event that can be completely captured by the practice. Referred by Li et al. (2009) as “cell storage intensities”, these values were 1.11 mm hr^{-1} and 0.59 mm hr^{-1} for the north and south sites, respectively. As shown in these figures, a variety of event magnitudes and durations led to occurrences of overflow at the north site (though overflow tended to be produced in higher-intensity storms), while larger magnitude rainfall events led to outflow production from the south site on a more consistent basis, presumably due to the addition of the underdrain system at this site which aids in elevating infiltration rate but also provides an additional outflow pathway. The majority of overflow production from the south site was largely associated during high-intensity rainfall events (Figure 5.4).

Similarly, the threshold event magnitude that produced outflow/overflow in each system can be determined by plotting rainfall depth (mm) against outflow/overflow volumes (m^3), as shown in Figures 5.5 and 5.6 (Winston et al., 2016). The discharge thresholds (D_t) for the north and south suspended pavement systems were determined to be approximately 9mm and 17mm, respectively. Because this threshold, which is well below the theoretical bowl storage capacity of the systems designed to contain runoff from the design storm event (25.4mm), corresponds solely to overflow production at the north site, it can be inferred that the available ponding volumes were not being completely utilized at the north practice. The distribution pipe network

Table 5.2: Summary of runoff partitioning in observed during the monitoring period.

	North Site		South Site	
	(mm)	(%)	(mm)	(%)
Inflow	1775	-	1887	-
Outflow	-	-	202	10.7
Overflow	3.3	0.2	11.4	0.6
Exfiltration/ET	1772	99.8	1673	88.7

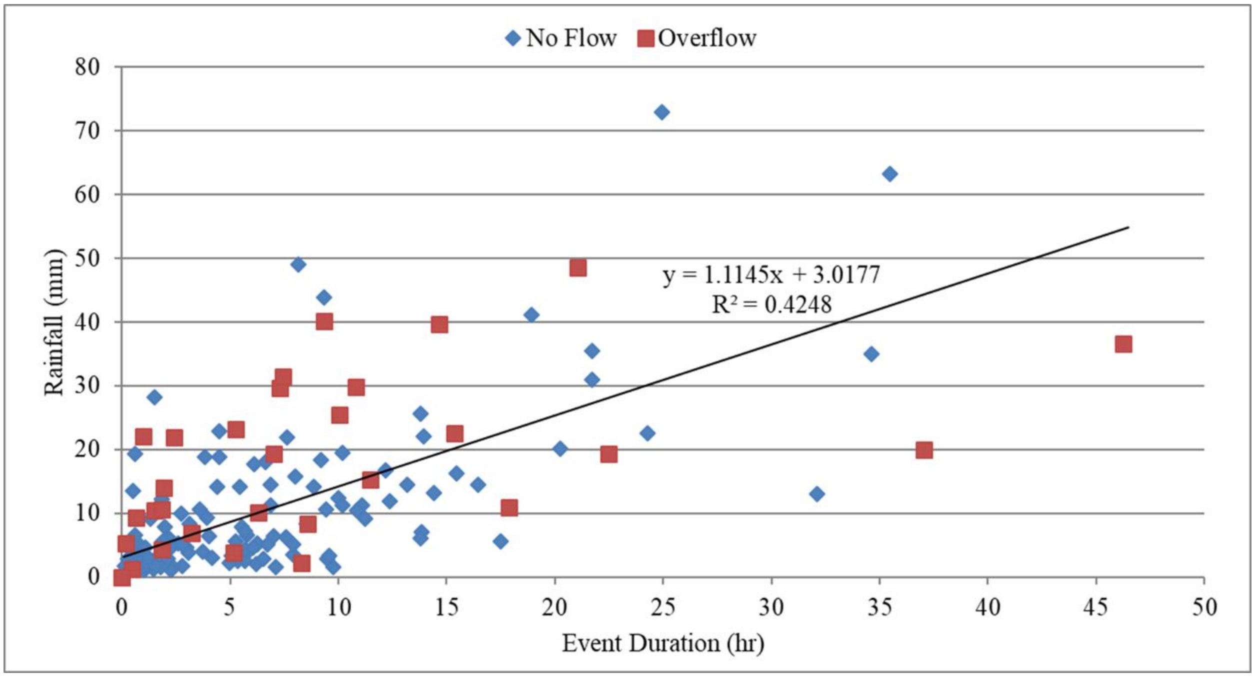

Figure 5.3: Influence of rainfall depth (mm) and duration (hr) on the occurrence of overflow for the north site. Regression line corresponds to rainfall-duration for events that did not produce overflows.

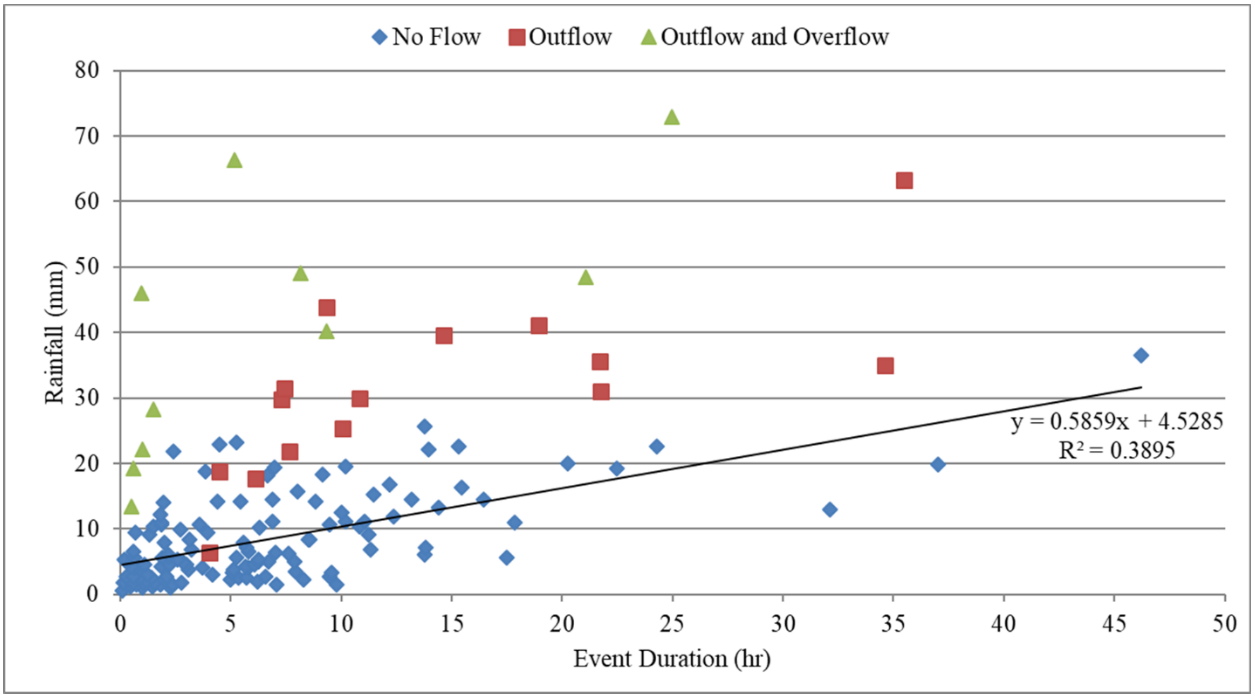

Figure 5.4: Influence of rainfall depth (mm) and duration (hr) on the occurrence of outflow and overflow for the south site. Regression line corresponds to rainfall-duration for events that did not produce outflows.

may have been overwhelmed at the north site during high-intensity storm events in the larger drainage area, causing overflow to occur before runoff had fully percolated into the bioretention storage area and thus resulting in a lower discharge threshold. That is, the hydraulics associated with the distribution system may have overshadowed the available ponding depth during some high-intensity storms. However, the significantly lower outflow/overflow volumes relative to inflow volumes suggest that both the north and south suspended pavement practices are effectively mitigating runoff volumes produced from both drainage areas. In particular, even when instances of overflow from the north site occurred, the amount of overflow was minimal.

The runoff volume reduction provided by the suspended pavement practices is comparable to values reported in previous bioretention literature. Research has demonstrated that runoff from small events is often completely captured by bioretention practices and produces no outflow to downstream waters because of soil porosity, exfiltration, and temporary bowl storage (Li et al., 2009, Davis et al., 2012). This trend was evident in the performance of the suspended pavement practices, which fully captured the majority of rainfall events that occurred during the monitoring period. When outflow-producing events do occur, studies have shown that bioretention practices can achieve significant volume reductions within the range of values observed in the suspended pavement practices. Examples of studies reporting mean volume reduction for bioretention practices within this range include: Selbig and Balster (2010) (95-100% runoff volume reduction), Brown and Hunt (2011) (87-100% runoff volume reduction), DeBusk and Wynn (2011) (97% runoff volume reduction), and Wadzuk et al. (2017) (89% runoff volume reduction).

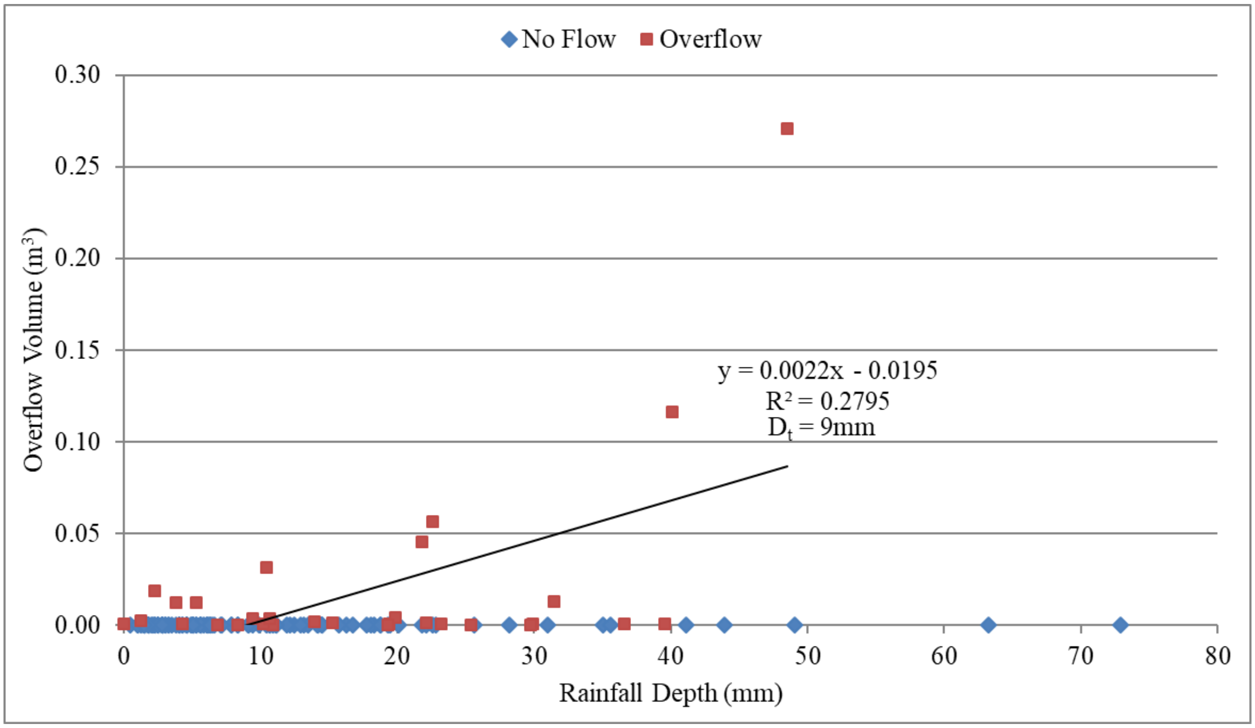

Figure 5.5: Discharge threshold (D_t) corresponding to overflow from the north site. Regression line describes overflow volumes and rainfall depths for overflow-producing events.

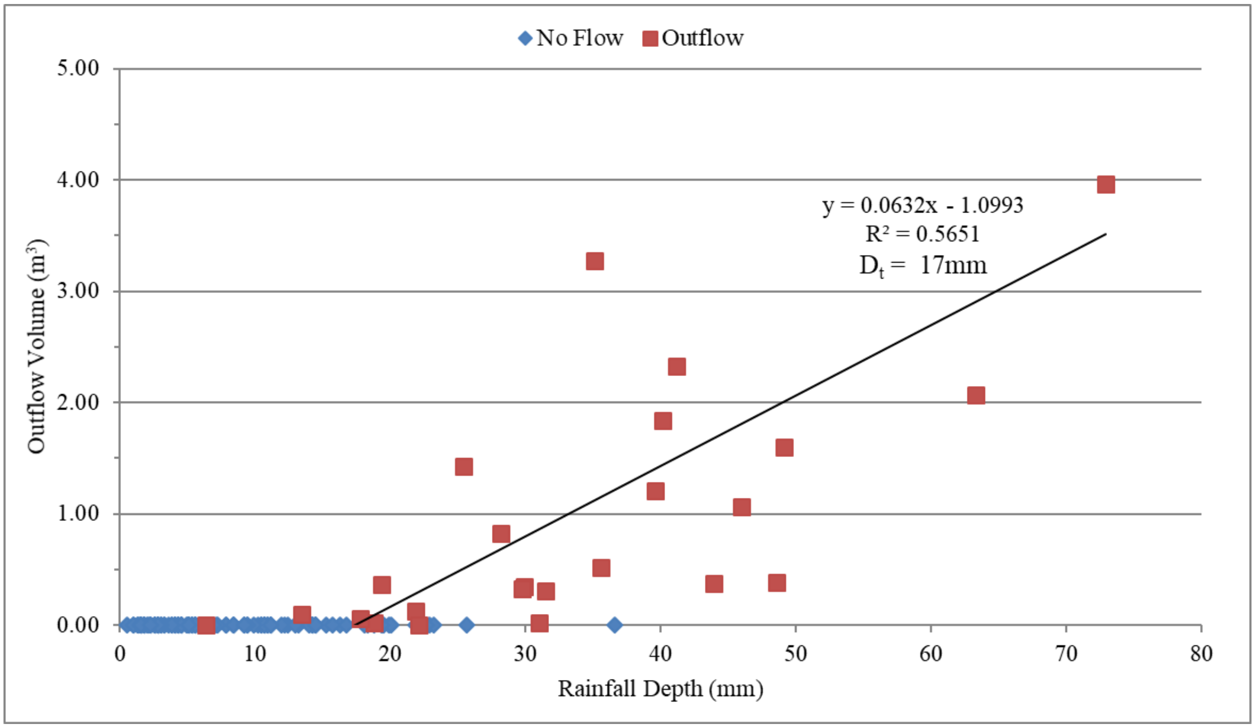

Figure 5.6: Discharge threshold (D_t) corresponding to the south site. Regression line describes outflow volumes and rainfall depths for outflow-producing events.

The suspended pavement practices studied herein exhibited greater runoff volume reduction compared to similar systems studied by Page et al. (2015) in Wilmington, North Carolina (USA). Page et al. (2015) reported that an estimated 20% of runoff volumes bypassed the Ann Street suspended pavement retrofit and 40% of rainfall events produced bypass from the site during a ten-month study period. In contrast, less than 1% of runoff was diverted to overflows from the north and south suspended pavement systems, while 20.5% and 8.1% of rainfall events resulted in overflow occurrences from the north and south sites, respectively. However, these discrepancies may be linked to differences in key design characteristics which likely impacted the function of these systems. The suspended pavement systems studied by Page et al. (2015) in Wilmington, NC were lined with an impermeable membrane for water quality sampling purposes, eliminating exfiltration from the practices. The Wilmington practices were also designed to include an internal water storage (IWS) layer, creating saturated conditions in half of the media used to fill the suspended pavement systems and promoting lower flow rates of water through the soil associated with soil saturation (Page et al., 2015). These sites also had one half of the available space between the top of the media and the bottom of the suspended pavement system decks (5cm) for temporary storage/subsurface ponding compared to the north and south sites in this study (10cm). Finally, the Ann Street retrofit studied by Page et al. (2015) had a much larger loading ratio (18:1) and thus received runoff from a greater contributing drainage area relative to the treatment area compared to the north and south suspended pavement systems (Table 5.1). The lack of exfiltration coupled with the presence of saturated media conditions, decreased available ponding volumes for temporary subsurface storage, and increased drainage area relative to the treatment area of the Wilmington suspended pavement practices likely account for the differences in hydrologic performance between the systems studied by Page et al. (2015) and the systems described herein. Though differences in performance exist between the two studies, the interaction between the bioretention media and surrounding soils found at the north and south suspended pavement practices may be more characteristic of unlined, freely draining practices, and may better reflect the hydrologic performance of suspended pavement systems in future non-research applications. These results also indicate that a larger catchment could have been treated by the north and south sites, suggesting the possibility that static volume design approaches may be highly conservative for suspended pavement systems.

Overall, these findings suggest that despite the limited ponded storage volume available in suspended pavement practices, the impact of these systems on runoff volume reduction is in-line with the established performance of bioretention practices more commonly installed in urban watersheds. Further, the water balances from the north and south suspended pavement system indicate that exfiltration into surrounding subsoils was a significant pathway for runoff volume reduction.

5.4.2 Water Quality

During the monitoring period, ten sets of paired inflow/outflow samples were collected from the south site (Table 5.3). The south suspended pavement system significantly reduced TSS levels, lowering the median influent TSS concentration of 167 mg L^{-1} to a median effluent of 6 mg L^{-1} (p<0.05). This result is unsurprising, given the ability of bioretention media to remove suspended solids from influent runoff reported in previous research (e.g., Hatt et al., 2009, Li and Davis, 2009, DeBusk and Wynn, 2011, Muha et al., 2016). No significant differences were observed between inflow and outflow samples for all nutrient and metal species, including Cu, Pb, Zn, PO_4^{3-}, NH_4^+-N, and NO_x-N. However, concentrations of Pb, NH_4^+-N, and effluent PO_4^{3-} present in all paired samples collected at the south site were below analytical detection limits, which limited the ability to interpret the pollutant removal performance of the system for these constituents after the study concluded.

While water quality improvements observed in this study were largely not significant, water quality samples collected from the suspended pavement systems studied by Page et al. (2015) showed consistently significant decreases in all pollutant concentrations from the inlet to the outlet of the practices. Similar to the previous discussion on system hydrology, differences in suspended pavement system pollutant removal performance may be attributable to design differences between the two studies. As with other studies that have reported the varying impact on IWS layers on NO_x-N removal (e.g., Dietz and Clausen, 2006, Hunt et al., 2006, Passeport et al., 2009, Brown and Hunt, 2011), the IWS layer included in the Wilmington systems studied by Page et al. (2015) likely improved NO_x-N removal rates comparted to the north and south suspended pavement systems described in this study, which were freely draining. Additionally, the impermeable membrane used in the Wilmington study may have maintained reduced conditions in the lower regions of the bioretention media between storms, leading to further reductions in NO_x-N concentrations in runoff held within the system before being flushed out in effluent during ensuing rainfall events (Page et al., 2015). The inclusion of an IWS layer in future suspended pavement systems may enhance pollutant removal, though further research is needed on unlined practices to confirm this hypothesis. In addition to design strategies, some differences in pollutant removal between the studies may be attributed to the effect of

Table 5.3: Median pollutant concentrations (st. dev.) collected from inflow and outflow of south suspended pavement system.

Pollutant	Inflow	Outflow
TSS (mg L^{-1})	**167 (69)**	**6 (21)**
NH_4^+-N (mg L^{-1})	0.01 (0.01)	0.01* (0.00)
NO_x-N (mg L^{-1})	0.05 (0.13)	0.11 (0.63)
PO_4^{3-} (mg L^{-1})	0.06 (0.03)	0.06* (0.00)
Cu (µg L^{-1})	0.5 (1.9)	0.3 (0.08)
Pb (µg L^{-1})	1.6* (0.0)	1.6* (0.0)
Zn (µg L^{-1})	7.9 (8.8)	7.9 (18.2)

Note: Bold font indicates significant differences in pollutant concentration (p<0.05). Asterisk (*) indicates that pollutant levels in all ten samples analyzed were below method detection limit (MDL).

environmental conditions, such as antecedent rainfall depth and temperature, which have been shown to influence effluent nutrient concentrations and bioretention performance (Manka et al., 2016).

In addition to the ten paired water quality samples, inflow samples were collected from 12 events that did not produce outflows (Table 5.4). Because their flows were completely diverted to exfiltration/ET, no pollutant loading from these storms was contributed to downstream waters. Thus, though the paired samples do not demonstrate significant water quality improvements for many of the constituents analyzed in the study (Table 5.3), the cumulative reduction in pollutant loads should be considered when evaluating the overall impact of the south suspended pavement system.

5.5 Conclusions and Recommendations

The use of suspended pavement systems provides stormwater engineers and urban foresters with an opportunity to institute green stormwater infrastructure in ultra-urban, space-limited areas while simultaneously promoting healthier urban trees. This study presented the hydrologic and pollutant removal performance of two suspended pavement systems designed to function as subsurface bioretention practices which were monitored over a period of 27 months. This work adds to the limited research of suspended pavement systems utilized in such applications. The free-drained north suspended pavement system diverted 99.8% of runoff to exfiltration/evapotranspiration and completely captured runoff from 79% of storms. Similarly, exfiltration/evapotranspiration accounted for 88.7% of runoff that entered the underdrained south site, which completely captured runoff from 83% of events during the study. Influent TSS concentrations were significantly reduced by the south suspended pavement system, though no

Table 5.4: Mean influent pollutant concentrations and load reductions ±SE for storm events (12) that did not produce outflows from the south suspended pavement system.

Pollutant	Inflow Concentration	Load Reduction
TSS (mg L^{-1} | kg)	148±39	40±10
NH_4^+-N (mg L^{-1} | g)	0.02±0.01	5.84±2.38
NO_x-N (mg L^{-1} | g)	0.08±0.04	16.44±7.44
PO_4^{3-} (mg L^{-1} | g)	0.06±0	14.00±0.00
Cu (µg L^{-1} | g)	1±0	0.23±0.08
Pb (µg L^{-1} | g)	2±0	0.36±0.00
Zn (µg L^{-1} | g)	12±2	2.85±0.38

other significant differences between influent and effluent pollutant concentrations were observed.

Further research is needed to add to the limited body of knowledge on these practices in order to improve the understanding of suspended pavement systems when used in stormwater management strategies. Future studies should investigate how design strategies (e.g., inclusion of IWS layers, altering media depth, loading ratios, etc.) influence the stormwater management performance of suspended pavement systems. Studies should also consider the treatment role provided by the trees planted in these systems, and how the unique characteristics of suspended pavement systems used in stormwater management applications (i.e., media composition, soil moisture regimes, urban pollutants, etc.), as well as the associated design variations noted above, influence the health of urban trees.

Last, the sizing criteria for these systems should be examined in future work. The design of the systems described herein were based on a static capture volume at the surface of the system (see NCDEQ, 2017). With the small ponding depth provided by suspended pavement systems, the loading ratios of the systems were high relative to literature, and the systems likely could have received substantially more water and remained effective. Consideration should be given to, at a minimum, allowing storage to be accounted for in the system sublayers (similar to TDEC, 2014 and MPCA, 2016). However, there is a minimum allowable soil volume for such systems to provide for tree health; thus, the amount of treatment area routed to a given practice must be optimized based on these two criteria.

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CHAPTER 6 : EVALUATING THE INFLUENCE OF DESIGN STRATEGIES AND METEOROLOGICAL FACTORS ON TREE TRANSPIRATION IN BIORETENTION SUSPENDED PAVEMENT SYSTEMS

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6.1 Abstract

Impervious surfaces, such as roads, parking areas, and buildings, found in cities throughout the world have significant impacts on urban hydrology due to increased volumes and peak flow rates of runoff delivered to receiving waterbodies. Bioretention practices are a common stormwater control measure (SCM) used to mitigate the impacts of urban runoff. When coupled with suspended pavement systems, which provide tree roots with an uncompacted soil matrix that enhances root access to oxygen and water, engineers can design subsurface alternatives to manage urban stormwater. Two suspended pavement systems designed to function as subsurface bioretention practices were installed on the campus of the University of Tennessee, Knoxville, Tennessee, USA. Sap flow sensors using the heat ratio method were installed in two bald cypress (*Taxodium distichum*) trees to characterize the role of transpiration in the suspended pavement systems. Mean transpiration rates were greater when water availability was higher in the bioretention media. Regression models indicated that atmospheric vapor pressure deficit (kPa) was the most influential environmental parameter on tree transpiration, and that stomatal regulation of water losses was evident when water was limiting. Findings from this study illustrate how tree transpiration rates can vary, even between individual trees of the same species, based on conditions within the practice and provide insight to practitioners on how design

parameters influence fine-scale tree-water relations in bioretention systems to maximize the contributions of transpiration on system hydrology.

6.2 Introduction

The prevalence of urbanization and the subsequent increases in stormwater runoff production have led to the deterioration of urban streams and receiving water bodies throughout the world (Walsh et al., 2005). With the evolution and enforcement of stormwater discharge-related regulations, cities and municipalities are implementing stormwater control measures (SCMs) as cost-effective, green infrastructure style approaches to refine their stormwater management programs (USEPA, 2009). An example of a type of green infrastructure implemented to manage urban stormwater are bioretention practices, one of the most popular and widely used SCMs in the United States and throughout the world (Davis et al., 2009). Bioretention practices typically consist of an excavated area of land backfilled with an engineered sandy soil media topped with mulch and various types of vegetation, though numerous design variations have been implemented in practice (Davis, 2008). In addition to improving water quality through biogeochemical treatment processes, a key objective of bioretention practices is the reduction of runoff volumes and peak flow rates to more closely mimic pre-development hydrology (Hunt et al., 2012). Bioretention designs influence stormwater runoff hydrology by utilizing soil media with relatively high infiltration rates (standards vary by state) and including bowl volumes for additional surface storage prior to infiltration (Davis et al., 2009). Volume reduction is primarily achieved through two mechanisms. First, runoff that has infiltrated into the soil media can exfiltrate into surrounding in-situ soils. Second, stormwater can be lost via soil evaporation and transpiration by vegetation, commonly combined as evapotranspiration (ET) (Berland et al., 2017). Previous research has demonstrated that ET can serve an important role in managing the water budget of bioretention practices.

Several studies have characterized the role of ET in bioretention practices through a variety of methods. Li et al. (2009) used a field-based water balance approach in a bioretention practice planted with unidentified trees and shrubs and lined with an impermeable membrane (to eliminate exfiltration) and found that losses due to ET accounted for 19% of runoff volume reduction. Winston et al. (2016) used DRAINMOD to model evapotranspiration from three low-permeability bioretention practices planted with a variety of plant types, including shrubs, sedges, native grasses, and trees, and reported that 4.5-5.5% of the water balance in each system could be attributed to ET. In a more controlled approach, Wadzuk et al. (2015) used weighing lysimeters to compare ET in bioretention mesocosms planted with native grasses and found that 50% of direct rainfall was converted to ET in freely draining systems, while mesocosms with an internal water storage (IWS) layer converted 78% of direct rainfall to ET, though the authors indicate these figures represent a high estimate. Similarly, Denich and Bradford (2010) used weighing lysimeters to measure ET in a bioretention practice and reported average ET rates of 4.2 mm d^{-1} in sunny, dry summer weather.

While these studies provide valuable insights on the overall water balance of bioretention practices, there is a relatively coarse understanding of ET and how it varies in these systems. Isolating transpiration from ET can provide critical information on temporal changes in this component of the water balance and ultimately improve plant selection and hydrologic modeling for bioretention systems. Further, literature is limited on the potential role of trees in bioretention practices. As a long-lived plant type with significant above-ground and below-ground biomass, trees may improve the hydrologic performance of bioretention practices (Scharenbroch et al., 2016). In one of the few studies of tree transpiration in bioretention practices, Scharenbroch et al. (2016) studied the impact of various tree species on the water balance of a parking lot outfitted with several green infrastructure practices. Using measurements of stomatal conductance to model monthly transpiration, transpiration from trees accounted for 46-72% of the total water outputs from the systems (Scharenbroch et al., 2016). However, differences in the responses of individual trees (both within and between species) to storm events may not be evident at the monthly timescale, and changes in transpiration patterns in response to varying design configurations were not investigated.

To address these knowledge gaps, transpiration rates of bald cypress trees (*Taxodium distichum*) planted in two field-scale bioretention suspended pavement practices installed on the campus of the University of Tennessee (Knoxville, TN USA) were studied using sap flow sensors. Sap flow sensors utilize high-resolution thermometric measurement techniques to relate the velocity of heat transfer through xylem tissue to tree water use (Burgess et al., 2001). Though measurements of sap flow have been widely implemented to characterize tree-water relations in forestry-related fields, no studies have conducted sap flow measurements in trees planted in bioretention practices to date. Average transpiration rates and the degree of influence of local meteorological conditions were compared between the two systems to evaluate the impact that bioretention function and differences in design parameters had on tree-water relations. The objective of this study was, for the first time, to utilize direct measurements of sap flow to quantify the impacts of hydrologic regime and design parameters on tree function and water use in bioretention practices.

6.3 Materials and Methods

6.3.1 Site Description

The study was conducted on the campus of the University of Tennessee (Knoxville, Tennessee, USA, 35.9606°N, 83.9207°W, elevation approximately 270m) between May and July 2017. The location is characterized by a temperate climate, with a mean annual temperature of 16°C and mean annual precipitation of 1215mm (Tennessee Climatological Service). Mean daily temperatures between May and July historically range from 19°C to 26°C in Knoxville, TN (National Weather Service). The study sites consisted of two suspended pavement systems that were installed in winter 2016. Suspended pavement systems are commercially available devices that transmit loads from paved surfaces to a compacted subsurface, creating a matrix of uncompacted soil media that promotes tree health by providing increased root access to air and water not commonly found in typical urban soils (Page et al., 2015). When backfilled with bioretention media, suspended pavement systems can become effective subsurface alternatives to traditional bioretention practices when space is limited, as is often the case in urban areas (Page et al., 2015). The two systems (hereafter referred to as the "north site" and "south site") were constructed in a similar manner, where an area of land, sized appropriately to the contributing drainage area using the Natural Resources Conservation Service (NRCS) curve number method and a design storm size of 25.4mm, was excavated, lined with a gravel subbase, and backfilled with bioretention media following the installation of the suspended pavement devices (USDA, 1986). The same suspended pavement system product and bioretention media were used in both sites. The drainage areas for both systems were completely impervious, with both sites receiving runoff from different sections of a small two-lane road. In both systems, stormwater runoff derived from the contributing drainage area was routed toward the inlet, where a series of perforated pipes distributed the water underground across the top of the bioretention media to percolate through the profile and allow for treatment. Runoff that had moved through the bioretention media at the north site drained into the underlying native soils, while an underdrain system was installed at the south site to collect and transmit excess stormwater that had percolated through the media profile (and did not exfiltrate into the underlying soils) to a monitoring point located at the back of the practice. Both sites were fitted with overflow pipe networks that bypassed the practices during extreme rain events. Due to local constraints, the systems were topped with topsoil and planted with turf grass in place of pavement. Bald cypress (*Taxodium distichum*) trees of approximately 5cm diameter at breast height (DBH) were planted in each system in March 2017. Root barrier devices were installed at both sites to direct the roots of the trees into the bioretention media. Due to its limited scope (i.e., only two unreplicated suspended pavement practices were studied), further research is needed to validate the results of this preliminary investigation into the performance of trees in suspended pavement systems, though findings from this work can provide some insights on the effect that design strategies have on tree performance in these practices. The cross-sectional components of the south suspended pavement system are shown in Figure 6.1. Additional information about the study sites is presented in Table 6.1.

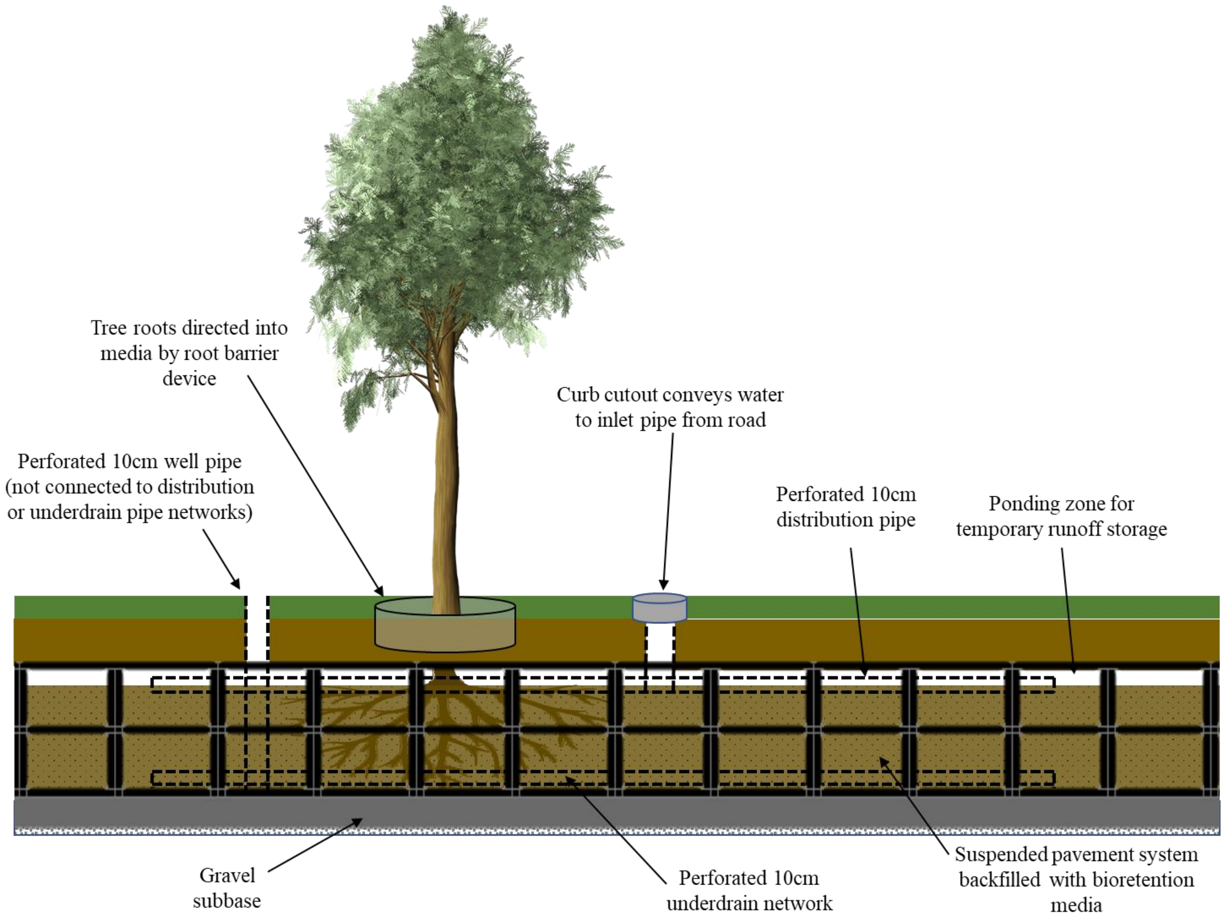

Figure 6.1: Cross-section showing components of south suspended pavement system.

Table 6.1: Description of bioretention suspended pavement systems.

Parameter	North Site	South Site
Drainage area (m^2)	183.0	138.5
Surface area (m^2)	22.3	27.5
Drainage configuration	No underdrain	Underdrain
Drainage area imperviousness (%)	100	
Depth of gravel subbase (cm)	10.2	
Media composition	93% sand, 7% fines, 5% organic matter (OM) by weight	
Media depth (cm)	71.1	

6.3.2 *Meteorological and Water Level Measurements*

Local climate data were collected at the study site using Campbell Scientific sensors taking temperature (T, °C), relative humidity, rainfall (P, mm), and total solar radiation (R_s, MJ m^{-2}) readings. Total solar radiation was assumed to be the equal at each site, though shading from adjacent vegetation may have caused potential differences in R_s. Weather data were recorded every minute and logged to a Campbell Scientific logger. The vapor pressure deficit (D, kPa) of the atmosphere was calculated using the ASCE Penman-Monteith method (Allen et al., 2005). Measurements of water depth in the suspended pavement systems relative to the bottom of the bioretention media were recorded every minute using UL-20 water level loggers (Onset HOBO) positioned in screened, perforated well pipes that were installed in each practice during construction. Local climate and water level data were collected for all 74 days of the study period.

6.3.3 *Sap Flow Measurements*

Sap flow measurements were performed using the heat ratio method (HRM) via SFM1 sap flow meters (ICT International) installed in each tree approximately 0.5m above the ground in mid-April 2017 (Burgess et al., 2001). The SFM1 sap flow sensors consist of a 35mm central heating probe abutted by two 35mm measurement probes, each containing two thermistors positioned 7.5mm and 22mm away from the tip of the probe to provide area-weighted measurements of sap flow radially across the sapwood. Measurements were conducted every 10 minutes by sending a pulse of heat from the central heating probe and recording the ratio of temperature increases at the upstream and downstream measurement probes over a fixed period. The rate at which this heat pulse travels up or down the stem can then be converted to sap flow using several equations and correction factors that account for wounding, probe misalignment, and water content of the sapwood. A more thorough explanation of measuring sap flow using the HRM can be found in Burgess et al. (2001). Because this study focused on comparing relative changes in tree water use between the two suspended pavement systems and destructive calibration techniques required to characterize sap flow (kg hr^{-1}) were not possible due to ongoing research, heat pulse velocities (V_h, cm hr^{-1}) were used as a proxy for transpiration, and the mean of the daily minimum readings at each site were used as zero-offset values for sensor calibration (Burgess, 2006). While measurements of V_h can provide qualitative insights on the patterns of tree water use, this approach cannot be used to directly quantify the amount of water used by trees. If quantifications of tree water use were desired, several calibration procedures and correction factors would be needed to transform readings of V_h in order to report rates of sap flow and tree water use. Due to the small DBH of the trees used in the study, the sapwood thickness was assumed to be equal to the diameter of the stem for both trees, similarly assumed in O'Brien et al. (2004). Sap flow data collection began approximately two weeks after sensor installation to account for the formation of wounds in the sapwood around the probes. Sap flow data were successfully collected for the entire 74-day study period at the south site, while 67 days of data were available for the north site due to equipment failures associated with depleted batteries used in the field to power the sensors.

6.3.4 Statistical Analysis

Meteorological data were compiled into daily and hourly means (T, D) or daily and hourly totals (P, R_s), while hourly means were used for water level and V_h readings. Inspections for normal distributions in the data were conducted using the Shapiro-Wilk test, which indicated that the water level and heat pulse velocity data collected at both sites were not normally distributed. Correlation analyses using Spearman's ρ were performed to identify connections between meteorological factors and both water level and sap flow. Wilcoxon Signed Rank tests were performed to test significant differences in sap flow values between sites. Time series and multivariable linear regression analyses were conducted to model the influence of daily averaged meteorological parameters (P, T, D, and R_s) on transpiration. Results from the Ljung-Box test indicated that significant autocorrelation was present in all parameters excluding temperature, so lag_1 terms were added to the models to reflect the influence of sap flow and meteorological conditions from the previous day on measurements of a given day. Models were created using mixed-direction stepwise regression techniques. Normality of the residuals was confirmed using the Shapiro-Wilk test, while autocorrelation in model residuals was assessed using the Durbin Watson test. All results were considered statistically significant at $p<0.05$. Statistical analyses were performed using the statistical software packages JMP Pro 13.2 (JMP, 1989-2007) and R (R Core Team, 2016).

6.4 Results and Discussion

6.4.1 Meteorological and Water Level Data

The meteorological data observed at the study site were characteristic of the typical transition between spring and summer in the Knoxville area (Figure 6.2). A total of 234.3mm of rain occurring on 33 of the 74 days of the study period was recorded. The daily mean temperatures ranged from a low of 9°C to a high of 28°C, with a mean daily temperature of 22.3°C occurring during the study. The mean daily vapor pressure deficit was 0.83kPa and the daily total solar radiation fluctuated between 6.7 MJ m^{-2} and 28.2 MJ m^{-2}. Water levels in the wells of both suspended pavement systems were significantly correlated with hourly rainfall, mean hourly D, and hourly total radiation readings ($p<0.01$). The mean water level in the north site during the study was 15.5cm, significantly higher than the mean south site water level of 8.2cm ($p<0.0001$, data not shown). Soil moisture sensors installed in each practice after the scope of this study support this trend, as mean soil water content was significantly higher in the north site than the south site ($p<0.0001$, data not shown). This indicates significant differences in the availability of water in the bioretention media profile at both sites, with the north site retaining more water and maintaining a higher water level than the south site, which may not have provided the bald cypress tree as much access to water due to the lower mean water depth. Differences in water level may be attributable to the presence of the underdrain network (or lack thereof in the north site), which likely regulated water levels in the south site, as well as potential variations in the composition of the subsoils underlying each suspended pavement system.

6.4.2 Sap Flow

Transpiration rates exhibited clear diurnal trends throughout the study, with maximum values near midday and minimum values occurring overnight (Figure 6.3). Heat pulse velocities at both sites were positively correlated to hourly mean T, D, and R_s, and negatively correlated to hourly P ($p<0.0001$). The interactions between these climate factors and tree physiological processes help explain the daily behavior of sap flow. Daily maximum sap flow values occurred during peak daily T, D, and R_s levels, as photosynthetic rates increased and water vapor concentration gradients between the leaf and the atmosphere reached peak values, followed by a decline in sap flow as these parameters decreased during nighttime hours (Pallardy, 2008). We hypothesize that sap flow declined following rain events as inundated, oxygen-deficient soils

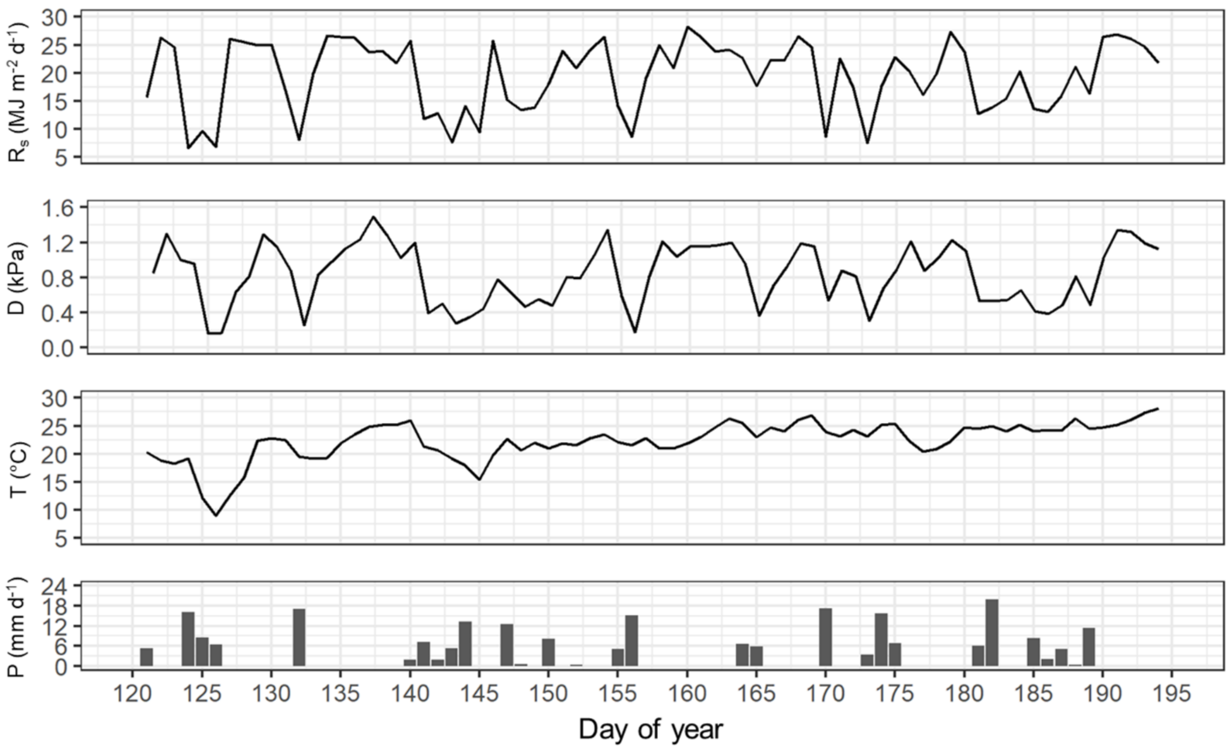

Figure 6.2: Trends of daily total solar radiation (R_s, MJ m^{-2}), mean daily vapor pressure deficit (D, kPa), mean daily air temperature (T, °C), and total daily rainfall (P, mm) observed at suspended pavement sites during study.

inhibited root respiration rates (Pallardy, 2008). Once runoff passed through the upper soil layers, soils became reoxygenated, and root respiration recovered, subsequent increases in sap flow rates occurred (e.g., as recorded between days 125-130 and 155-160) (Figure 6.3). An example of how water level and heat pulse velocities were influenced by a rainfall event is shown in greater detail in Figure 6.4.

The mean hourly heat pulse velocity at the north site (2.65 cm hr^{-1}) was significantly higher than the mean value recorded at the south site (2.38 cm hr^{-1}) ($p<0.0001$). When considering the difference in water availability between the two sites, these results are consistent with previous research where sap flow sensors were employed to characterize tree transpiration, along with other studies of ET in bioretention practices. Gazal et al. (2006) reported higher total annual transpiration in cottonwood trees grown alongside a perennial stream, where water was not limiting, than others grown near an intermittent stream, where trees exhibited greater signs of water stress due to lack of water availability. Berland et al. (2017) suggest that sustaining high ET rates in green infrastructure practices require adequate soil moisture levels to be maintained. Relating these observations to bioretention, results from Wadzuk et al. (2015) indicated higher portions of the water balance of bioretention practices with IWS layers that maintained soil moisture could be attributed to ET. The significant differences in transpiration rates observed in the suspended pavement systems provide further evidence that maintaining a region of high soil moisture within the bioretention media profile, whether it is derived from the inclusion of an IWS layer or due to lower permeability subsoils surrounding the practice, can influence the water balance through increased transpiration rates, even between individuals of the same tree species.

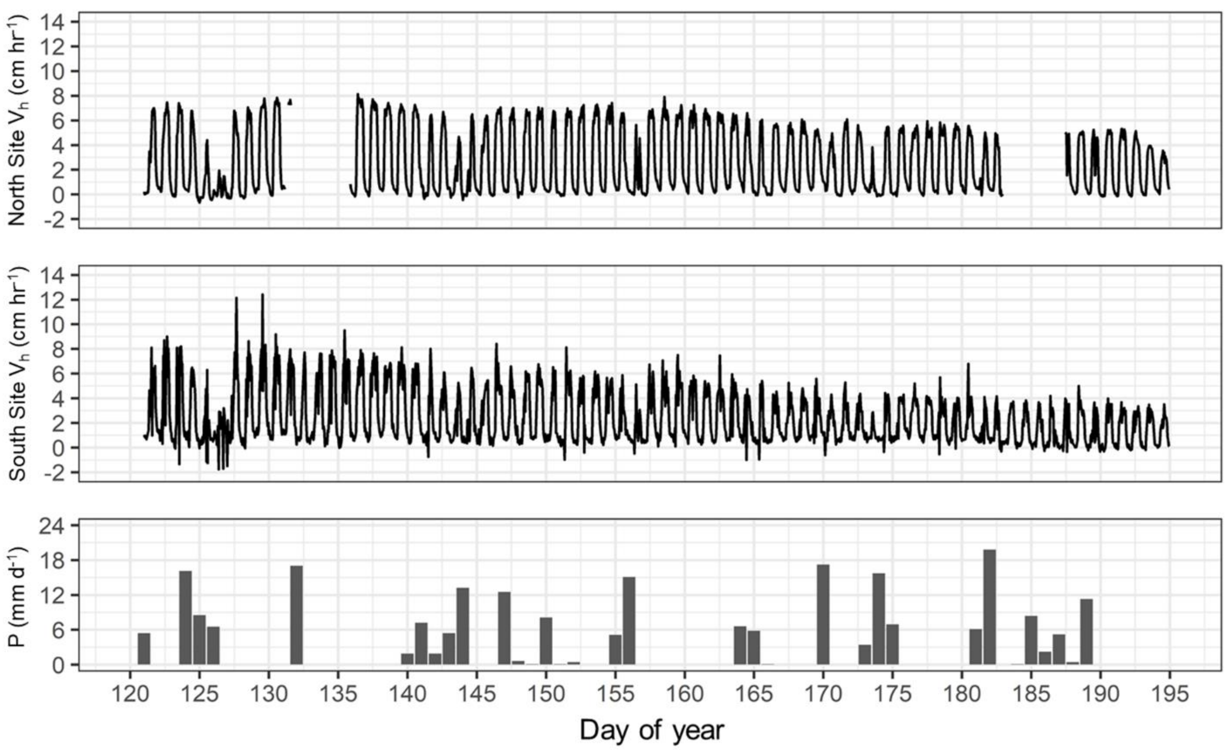

Figure 6.3: Diurnal trends of mean hourly heat pulse velocity (cm hr^{-1}) and daily rainfall totals (mm d^{-1}) recorded during the study. Missing heat pulse velocities at the north site occurred due to equipment power failure.

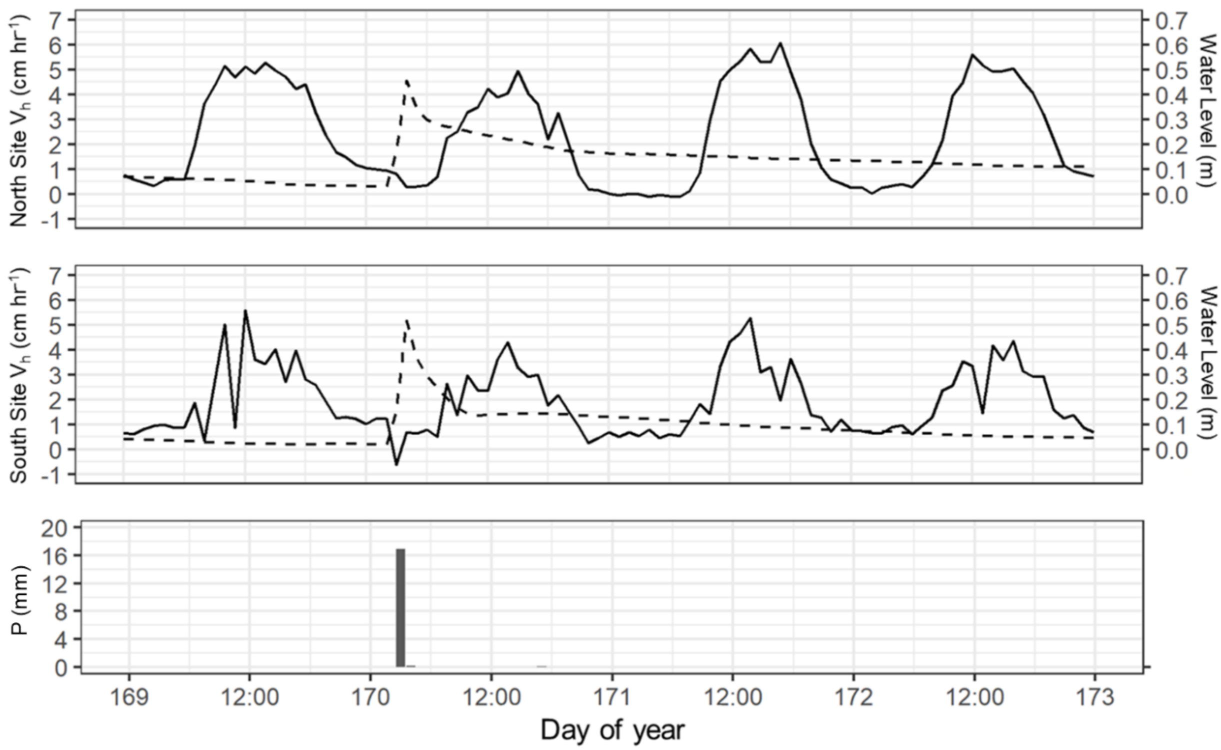

Figure 6.4: Trends in heat pulse velocities (solid line) and water level (dashed line) at suspended pavement sites following a 17mm rain event on day 170. Peak heat pulse velocities at both sites declined from their relative maximum values on day 169 and recovered on day 171 as water levels in the systems receded due to drainage.

6.4.3 *Environmental Influences on Transpiration*

One potential outlier (on day 136) was identified in the residual-predicted plot for the north site regression model. Due to the lag terms used in the model, this reading was likely influenced by the equipment power failure that occurred from day 132-135 at the north site, which produced a subsequently low mean daily sap flow on day 135 after power was restored. Therefore, this point was removed from the regression model for the north site, which improved model prediction accuracy and eliminated autocorrelation from the residuals. No such outliers were identified for the south site. Parameters determined to be significant in the stepwise regression analyses using meteorological factors to model sap flow are shown in Table 6.2.

Regression models for both suspended pavement systems effectively predicted tree transpiration using daily averaged meteorological parameters ($p<0.0001$). The model results suggest that vapor pressure deficit (D and lag D) and antecedent heat pulse velocity (lag V_h) explained the most variability in transpiration rates. While not significant for the north site model, transpiration rates at the south site were significantly explained by mean daily air temperature of the previous day (lag T), though the regression coefficient was small, indicating that changes in air temperature did not produce a large change in V_h. The positive relation between D and V_h is not surprising, as evaporative losses spurred by vapor pressure differentials between the leaf and the atmosphere constitute the majority of water used by trees (Pallardy, 2008). This finding agrees with results from Chen et al. (2011), who also found that transpiration in urban trees in China was significantly controlled by D. The negative influence of antecedent D and V_h on transpiration levels in the models is also reasonable. High atmospheric water demand and correspondingly high transpiration rates limit the ability of the tree to rehydrate at night via

Table 6.2: Results of regression analyses using daily averaged meteorological values (lag D, lag T) and antecedent heat pulse velocities (lag V_h) as independent variables to model sap flow (V_h). Lag terms correspond to conditions from the previous day. All model parameters were significant at $p<0.01$.

Model Parameter	North Site	South Site
D, kPa	1.80	1.35
Lag D, kPa	-1.60	-1.06
Lag T, °C	-	-0.05
Lag V_h, cm hr^{-1}	0.80	0.77
Intercept	-	1.14
Final Model	V_h = 1.80*D – 1.60*lag(D) + 0.80*lag(V_h)	V_h = 1.35*D – 1.06*lag(D) – 0.05*lag(T) + 0.80*lag(V_h) + 1.14
R^2	0.79	0.80

water absorption through roots, creating decreased water availability for transpiration during the following day.

Though the regression models suggest that the same environmental factors influenced transpiration rates in both sites, the magnitude of the model parameters indicate that the response of V_h to environmental changes was different between the suspended pavement systems. Atmospheric moisture conditions had a greater influence on transpiration rates in the north site, as changes in D and lag D produced 33% and 51% larger responses in V_h in the north site compared to the south site, respectively. With the differences in water availability between the sites established via the water level measurements, these results suggest that stomatal regulation to limit water losses is occurring in the tree in the south site, while the higher water availability in the north site makes this water conservation strategy less necessary. This conclusion again aligns with findings from Gazal et al. (2006), who suggested that a lack of significant relationship between sap flow and vapor pressure deficit in trees growing in water-limited conditions (i.e., near intermittent streams) indicated the influence of stomatal control on water losses. Conversely, the authors of this study found this relationship to be significant in trees growing in non-water limiting environments (i.e., along perennial streams), indicating a low resistance to water losses (Gazal et al., 2006). These restrictions to transpiration responses to vapor pressure deficit likely resulted in the lower mean heat pulse velocity rate observed in the south site.

These findings may have important design implications when characterizing the influence of transpiration on bioretention hydrology. Unlike most urban settings, where characterizing tree water use is important from a resource conservation perspective (such as mitigating transpiration losses of water used for irrigation), stormwater engineers may often seek to maximize transpiration losses in bioretention practices when volume reduction is a key runoff management objective. These results suggest that volume reduction via transpiration may be reduced in a scenario where water is limiting due to stomatal regulation of water losses compared to site conditions characterized by high soil moisture and water availability, even among trees of the same species. It should be noted that prolonged saturated soil conditions impair tree function, even in highly flood tolerant species. Therefore, trees are likely to contribute more to volume losses via transpiration when higher, though not saturated, soil moisture conditions are maintained in bioretention practices, which can be achieved through design strategies such as the inclusion of internal water storage (IWS) layers or allowing runoff to slowly percolate into underlying subsoils instead of underdrain networks (when applicable).

6.5 Conclusions and Recommendations

Transpiration rates of two bald cypress (*Taxodium distichum*) trees grown in suspended pavement systems backfilled with bioretention media were studied using sap flow sensors during the summer of 2017. During the study, transpiration rates were dependent on site conditions that influenced water availability, and tree responses to environmental conditions were varied. Lower transpiration rates were observed in the more water-limited conditions of the underdrained system. Further, regression models showed that atmospheric vapor pressure demand significantly influenced transpiration rates, and that water availability was connected to the degree of stomatal control on water loss. Results from this study suggest that transpiration losses are greater in bioretention systems with higher water availability. Therefore, to maximize the contribution of transpiration on runoff volume reduction, stormwater engineers should consider design strategies that promote these conditions while minimizing prolonged saturation in the upper soil layers, such as the inclusion of an internal water storage layer or promoting runoff percolation into surrounding soils in place of a perforated underdrain network, when appropriate. Future research should investigate other tree species to identify potential differences in their ability to regulate water losses when water limiting conditions exist.

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CHAPTER 7 : CONCLUSIONS AND RECOMMENDATIONS FOR FUTURE WORK

7.1 Summary and Conclusions

This dissertation has explored the role of urban trees in bioretention practices through field assessments of tree health, a mesocosm-scale laboratory study, and a performance evaluation of field-scale research installations. As discussed in Chapter 3, the health of trees in bioretention systems in Tennessee and North Carolina were characterized and compared to other urban trees, and random forest regression models were constructed to identify parameters relating to the bioretention growing environment that influence tree health. Results from the comparison of tree health showed that trees exhibited greater health when their natural preferences for growing condition were similar to the conditions expected to be found in bioretention practices (i.e., sandy, well-drained, nutrient-deficient soils with soil moisture regimes characterized both periods of inundation and drought). The random forest models described in this chapter revealed that a particular subset of environmental parameters related to bioretention media composition, media chemistry, and species selection and planting location were most influential to tree health in bioretention practices. Based on the results of this study, it was determined that the health of trees in bioretention practices may be improved if species selection is informed by an analysis of bioretention media composition and chemistry and the degree of compatibility between the unique growing conditions found in bioretention and the preferred growing conditions of the species. In doing so, designers may be able to simultaneously improve the performance of bioretention systems while promoting tree health, thus increasing the overall impact of the practice through the ancillary environmental benefits provided by healthy urban trees (e.g., mitigation of the urban heat island effect, improved air quality, reduced noise pollution, etc.).

The mesocosm-scale study of tree contributions to bioretention practices presented in Chapter 4 showed that trees provide significant contributions to bioretention performance and that differences between various tree species and their role in these practices exist. Differences in effluent pollutant concentrations from the various mesocosm configurations were generally not significant, indicating the dominant role of the bioretention media in pollutant removal. However, as the trees mature and their roots occupy a greater relative volume of soil, it can be expected that plant uptake would represent a more significant pollutant removal pathway in bioretention practices. Evapotranspiration rates from mesocosms planted with trees were significantly higher than nonvegetated mesocosms, highlighting the role of transpiration, which accounted for between 8.2-37.5% of average daily water losses and served as a significant hydrologic pathway in the mesocosms. It was determined that species with the highest degree of plant development, canopy size, and growth exhibited the greatest hydrologic impact via evapotranspiration, further emphasizing the importance of appropriate species selection to maximize the contributions of trees to bioretention performance.

The results of a 27-month hydrologic and water quality monitoring effort of two bioretention suspended pavement systems were presented in Chapter 5. The majority of the water balance at both systems was comprised of exfiltration and evapotranspiration, while outflow and/or overflow volumes produced from the practices were minimal. The free-drained north system diverted 99.8% of runoff to exfiltration/evapotranspiration, which also accounted for 88.7% of runoff that entered the underdrained south site. However, occurrences of outflow and overflow from the systems were infrequent, as 79% and 83% of storm events were completely captured by the north and south sites, respectively. Results from water quality sampling conducted at the south system showed significant reductions in influent TSS concentrations, though significant removal of other pollutants was not observed. Findings from this study confirmed the ability of suspended pavement practices to mitigate runoff volumes and the viability of these systems when used in urban stormwater management applications.

Results from the in-situ study of tree transpiration in bioretention suspended pavement systems in Chapter 6 demonstrated the impact that design strategies and local weather conditions have on tree transpiration. Sap flow sensors installed in each of the bald cypress (*Taxodium distichum*) trees planted in the practices showed that tree transpiration rates responded directly with water availability in the bioretention media. Regression models showed that transpiration was significantly influenced by vapor pressure deficit and that stomatal regulation of water losses were occurring when water availability was limited. Results from this study suggest that transpiration may be a more significant hydrologic pathway in bioretention systems when water availability is greater. In order to maximize the role of transpiration in bioretention hydrology, it was recommended that design strategies which promote these conditions while minimizing prolonged saturation in the upper soil layers, such as the inclusion of an internal water storage layer or promoting runoff percolation into surrounding soils in place of installing an underdrain network, be instituted when conditions allow.

7.2 Limitations and Recommendations for Future Work

Though this research identified factors that promote healthier trees in bioretention practices and characterized the contributions they provide to urban stormwater management, further research is needed to more fully understand the role of trees in bioretention and the criteria on which tree species should be selected to maximize bioretention functionality. The geographic scope of the field assessment of tree health in Chapter 3 was limited to bioretention practices in the inland southeastern United States, which may experience different urban stormwater challenges than green infrastructure in other regions of the country. There were also a limited number of both trees and tree species observed in this study, which may not be representative of the health of other species in bioretention practices outside of this region. Though the trees selected for the mesocosm-scale study in Chapter 4 represented a diverse array of species native to the study region, the size and age of the trees used in the study (i.e., two-year old seedlings) were constrained by the scope of the study. While it was determined that the trees provided significant contributions to the hydrology of the mesocosms, the size of the trees (and the relatively small volume of soil occupied by their roots) may have limited the potential impact of plant uptake on pollutant removal from the systems.

The performance study of the suspended pavement systems presented in Chapter 5 investigated a limited number of design variations in these versatile practices. Though the presence of an underdrain (or lack thereof) influenced hydrology, alternative variations in design (such as the inclusion of an internal water storage layer) were not investigated. Further, the pollutant removal capabilities of the south system may not have been fully captured by the number of storms from which water quality samples were collected. Finally, the scope of the in-situ study of tree transpiration in bioretention suspended pavement practices (Chapter 6) introduced limitations relating to the number of species that were investigated, the maturity of the trees used in the study, the duration of the study, and the number of replications analyzed in the results.

With these limitations in mind, the following recommendations for future research are proposed:

- Studies should expand upon the number of trees, the diversity of species, and the climate regime associated with the geographic region examined in this research to provide a more thorough investigation of tree health in bioretention with the goal of identifying any potential physiological traits common to successful species in bioretention. Furthering the understanding of the environmental factors that are most influential to tree health and the way these factors impact different species may provide additional insights to stormwater engineers and urban foresters on how to optimize species selection in future bioretention practices.
- Studies should characterize the role of trees in bioretention throughout their lifespan, investigating the way that hydrologic and pollutant removal contributions to bioretention practices change over various time periods associated with tree growth cycles (e.g., seasonally). Though individual tree pit installations are common, studies should investigate the potential interactions between trees and other plant types found in bioretention practices (e.g., shrubs, grasses, sedges, etc.), and how planting plans composed of trees and other plant types influence bioretention performance.
- Research should add to the limited body of knowledge on the performance of suspended pavement practices in stormwater management applications. Studies should seek to characterize the treatment provided specifically by trees in these systems, and how

variations in design and tree characteristics (i.e., species, age) influence tree contributions to stormwater management.

- Studies should increase the use of in-situ measurements of tree transpiration in bioretention practices, along with the diversity and number of tree species investigated herein, to further the understanding of the role of transpiration in bioretention hydrology. Studies utilizing these measurement techniques should consider various design strategies to characterize the response of transpiration patterns to the changes within the bioretention environment imparted by design configurations.

APPENDICES

Appendix A: Random Forest Algorithm and R Code

Random Forest Regression Algorithm

```
while number of parameters in model is greater than zero do
  build RF regression model
  identify the least important parameter
  for each tree in the dataset do
    exclude the tree from the dataset and
     build RF regression model for remaining trees in the dataset
    use the RF model to estimate the response variable of the
     excluded tree
    calculate the difference between the estimated response
     variable and the ground truth (i.e., error)
  end for
  calculate cross validation error by averaging all errors
  remove the least important parameter in model
end while
```

Figure A.1: Random forest regression algorithm.

Random Forest R Code for Composite Crown Volume (CCV)

```
mydata = read.csv(file.choose())
attach(mydata)
parameters = names(mydata[,3:21])
n=length(parameters)

CCV.df = data.frame(matrix(ncol=58, nrow=0))
names(CCV.df)[1] = c("x")
names(CCV.df)[2:20] = paste("Var", n:1, sep="")
names(CCV.df)[21:39] = paste("delta", n:1, sep="")
names(CCV.df)[40:58] = paste("ave.delta", n:1, sep="")

for (j in 1:25){
print(j)
df = mydata[,3:21]

var.elim = data.frame(matrix(ncol=n, nrow=1))
names(var.elim)[1:n] = paste("Var", n:1, sep="")
```

```
delta = data.frame(matrix(ncol=n, nrow=1))
names(delta)[1:n] = paste("delta", n:1, sep="")

ave.delta = data.frame(matrix(ncol=n, nrow=1))
names(ave.delta)[1:n] = paste("ave.delta", n:1, sep="")

for (i in 1:n){
  Tree.rf = randomForest(df, CCV, ntree=25000, importance=T)
  imp.df = data.frame(importance(Tree.rf))
  imp.df = imp.df[order(imp.df$X.IncMSE),]

  predicting.fields = names(df)

cross.validation = function(mydata, predicting.fields)
{

  response = data.frame(matrix(ncol=5, nrow=0))
  names(response) = c("Fold","TreeNo","pred","actual","delta")

  all.fields = c("TreeNo","CCV", predicting.fields)
  fold = 0
  for(tree in unique(mydata$TreeNo))
  {
    fold = fold+1
    testing = c(tree)

    training.set = subset(mydata, !(TreeNo %in% testing), select=all.fields)
    testing.set = subset(mydata, TreeNo %in% testing)

    rf = randomForest(CCV~., data=training.set[2:ncol(training.set)], ntree=25000,
importance=TRUE)

    pred = predict(rf, testing.set[3:ncol(testing.set)], type="response")

    x = c(as.character(pred[[1]]),as.character(testing.set[,2]))
    response = rbind(response, data.frame(Fold=fold, TreeNo=tree, pred=x[1], actual=x[2]))
  }
  return(response)
}

response = cross.validation(mydata,predicting.fields)
response$delta = abs(as.numeric(as.character(response$pred))-
as.numeric(as.character(response$actual)))
```

```
if (i==n){
  updated.imp.df = imp.df[1:dim(imp.df)[1],]
  updated.fields = labels(updated.imp.df)[[1]]
  var.elim[1,i] = labels(imp.df[1,])[[1]]
}

else {
  updated.imp.df = imp.df[2:dim(imp.df)[1],]
  updated.fields = labels(updated.imp.df)[[1]]
  var.elim[1,i] = labels(imp.df[1,])[[1]]
}

df = subset(df, select=c(updated.fields))

delta[1,i] = sum(response$delta)
ave.delta[1,i] = delta[1,i]/nrow(mydata)
print(delta)
print(var.elim)
}

total1 = merge(j,var.elim)
total2 = merge(total1,delta)
total3 = merge(total2,ave.delta)
CCV.df = rbind(CCV.df,total3)
print(CCV.df)
}

#Final Dataframe####
print(CCV.df)
```

Appendix B: Mesocosm Evapotranspiration Analysis

Matlab Code for Data Smoother

```
clear all;
scaledata = csvread('rawscaledata.csv');
N = length(scaledata);

Fs = 24*60

delta_t = 1/Fs;
t=0:delta_t:delta_t*(N-1);
plot(t,scaledata);
xlabel('Time (days)');
ylabel('Amplitude (l/s)');

Y = fft(scaledata);
f_delta = (Fs/N);
f=0:f_delta:f_delta*(N-1);

plot(f(1:N/2),abs(Y(1:N/2)));
xlabel('Frequency (1/day)');
ylabel('Amplitude');

[b,a] = butter(5,.0018,'low');
[h,f] = freqz(b,a,floor(N/2),Fs/2);

plotyy(f(1:N/2),abs(Y(1:N/2)),f(1:N/2),abs(h(1:N/2)));
xlabel('Frequency (1/day)');
ylabel('Amplitude');

scaledata_filtered = filtfilt(b,a,scaledata(1440:end));
plot(t,scaledata,t(1440:end),scaledata_filtered,'r');
xlabel('Time (days)');
ylabel('Amplitude (l/s)');

csvwrite('filename.dat',scaledata_filtered)
```

Identification of Evapotranspiration Start Point

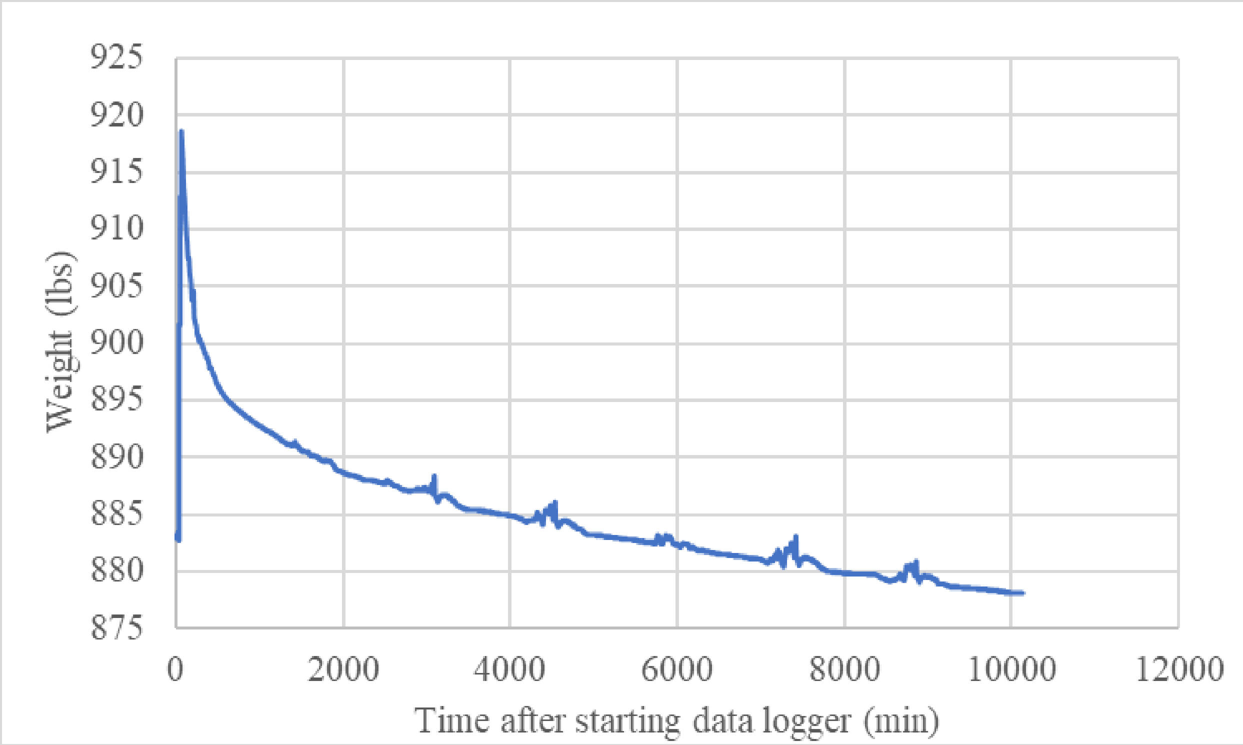

Figure B.1: Raw scale data from Pin Oak 3 from July 31, 2017 to August 7, 2017.

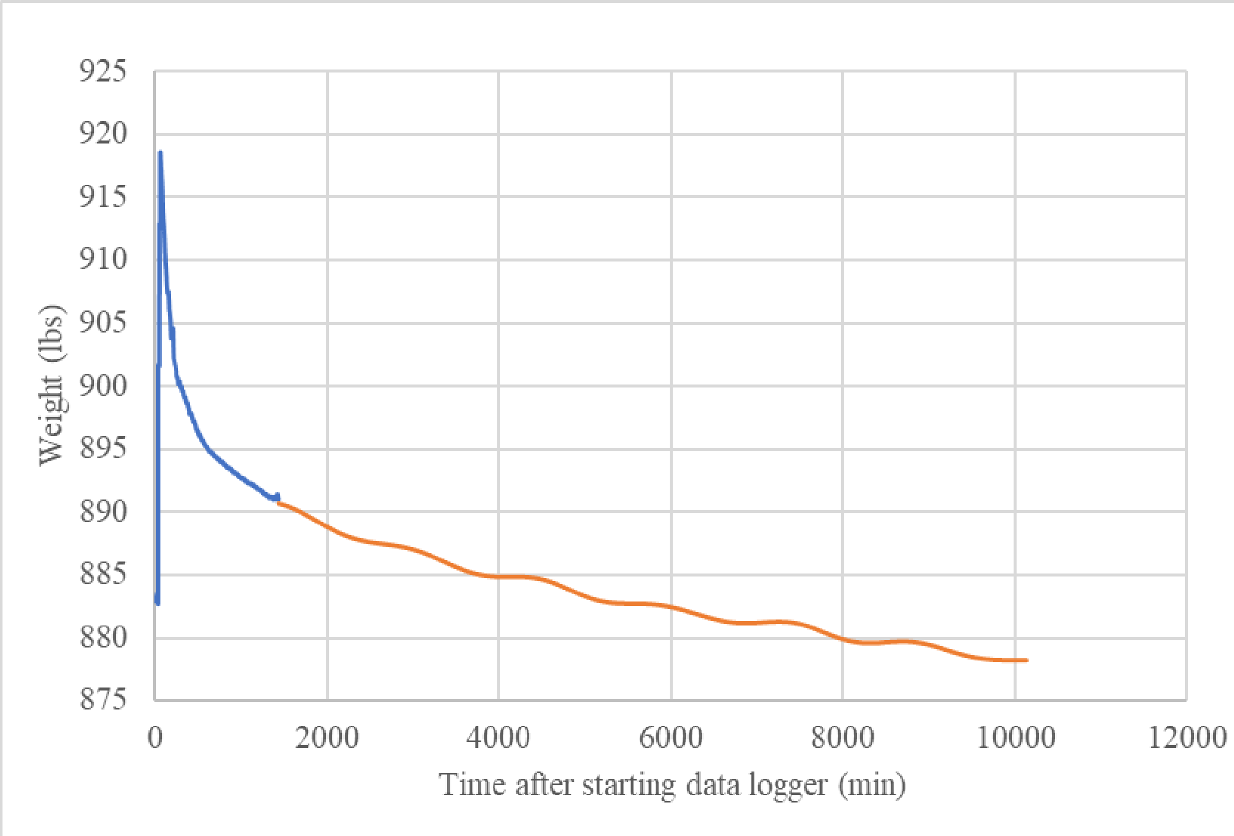

Figure B.2: Raw scale data (blue) and smoothed data applied to readings 24hr after starting logger (orange).

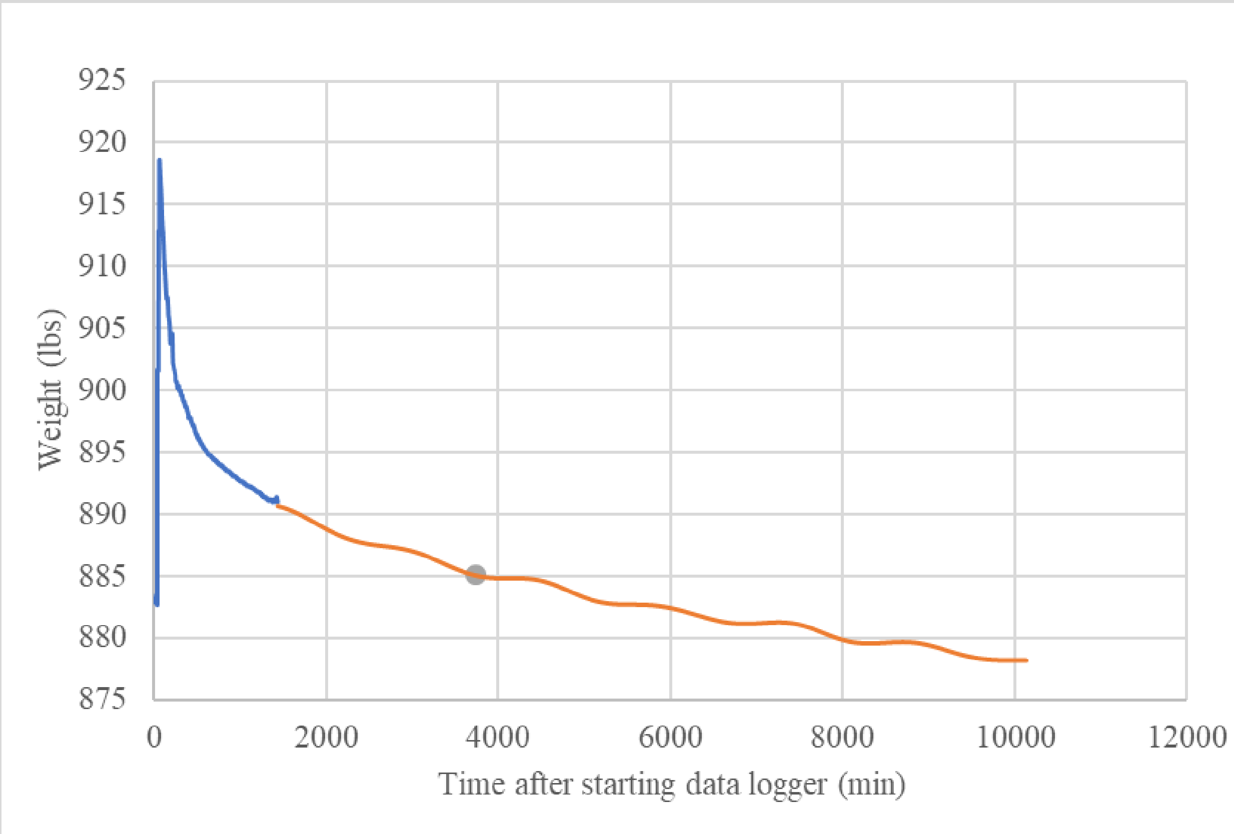

Figure B.3: Raw scale data (blue), smoothed data (orange), and start point of evapotranspiration (grey).

Table B.1: Determination of evapotranspiration rate for Pin Oak 3 from August 3, 2017 to August 7, 2017.

Data Point	*Time*	*Weight (lbs)*
ET Start Point	August 3, 2017 2:00AM	885.2
End of Dry Period	August 7, 2017 12:34 PM	878.1
Total ET Weight Loss		7.1
ET Rate		**2.2 mm d^{-1}**

Appendix C: Suspended Pavement System Site Photos

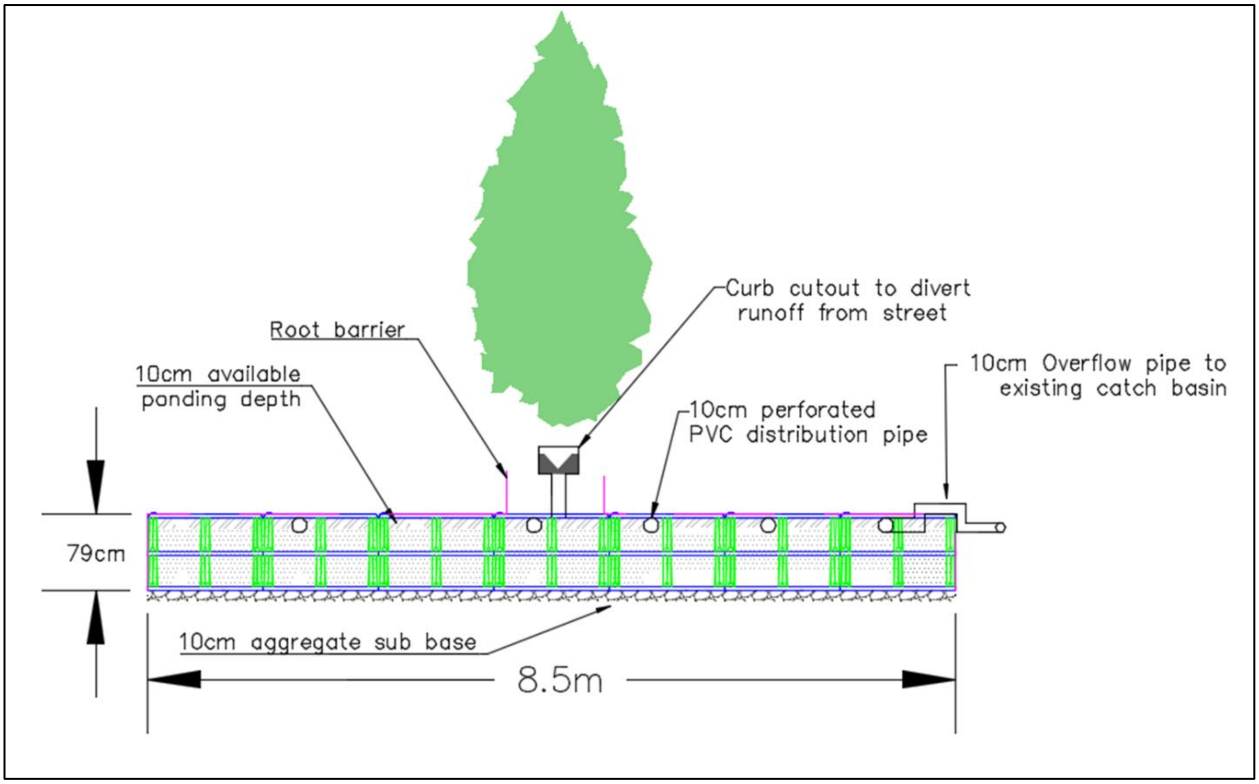

Figure C.1: Cross-section showing design components at north suspended pavement site.

Figure C.2: Installation of silva cell (Deeproot) frames and underdrain network at south suspended pavement site.

Figure C.3: 0.4’ HS-Flume installed at inlet of suspended pavement systems.


Figure C.4: SFM1 sap flow sensor (ICT International) installed in trees planted in suspended pavement systems.

Appendix D: Summarized Hydrologic Data Collected from Suspended Pavement Systems

Table D1: Summarized hydrologic data collected from suspended pavement systems.

DATE	PRECIPITATION			SOUTH INFLOW	SOUTH OUTFLOW	SOUTH OVERFLOW	NORTH INFLOW	NORTH OVERFLOW
	Depth (in)	Depth (mm)	Duration (hrs)	(m^3)	(m^3)	(m^3)	(m^3)	(m^3)
04-06-16	0.74	18.80	3.83	2.57	0.00	0.00	3.40	0.00
04-12-16	0.21	5.33	2.58	0.71	0.00	0.00	0.93	0.00
04-22-16	0.72	18.29	9.17	2.50	0.00	0.00	3.30	0.00
04-28-16	0.17	4.32	1.83	0.57	0.00	0.00	0.75	0.00
04-30-16	0.89	22.61	15.33	3.10	0.00	0.00	4.09	0.06
05-02-16	0.86	21.84	2.42	2.99	0.00	0.00	3.95	0.05
05-12-16	0.33	8.38	8.58	1.13	0.00	0.00	1.49	0.00
05-20-16	0.52	13.21	14.42	1.80	0.00	0.00	2.37	0.00
05-29-16	0.36	9.14	1.33	1.23	0.00	0.00	1.63	0.00
06-01-16	0.43	10.92	1.83	1.48	0.00	0.00	1.96	0.00
06-02-16	0.11	2.79	0.58	0.35	0.00	0.00	0.47	0.00
06-24-16	0.90	22.86	4.50	3.13	0.00	0.00	4.14	0.00
07-05-16	0.49	12.45	10.00	1.69	0.00	0.00	2.23	0.00
07-26-16	0.07	1.78	0.15	0.22	0.00	0.00	0.28	0.00
07-28-16	0.18	4.57	2.24	0.60	0.00	0.00	0.79	0.00
07-29-16	0.02	0.51	0.08	0.05	0.00	0.00	0.06	0.00
08-02-16	0.21	5.33	0.17	0.71	0.00	0.00	0.93	0.01
08-04-16	0.31	7.87	5.56	1.06	0.00	0.00	1.40	0.00
08-08-16	0.37	9.40	0.67	1.27	0.00	0.00	1.68	0.00
11-19-16	0.18	4.57	1.08	0.60	0.00	0.00	0.79	0.00
11-24-16	0.07	1.78	2.77	0.22	0.00	0.00	0.28	0.00
11-28-16	0.64	16.26	15.47	2.22	0.00	0.00	2.93	0.00
12-17-16	0.16	4.06	5.82	0.53	0.00	0.00	0.70	0.00
01-10-17	0.57	14.48	13.18	1.97	0.00	0.00	2.61	0.00
01-13-17	0.11	2.79	9.47	0.35	0.00	0.00	0.47	0.00
01-15-17	0.04	1.02	1.02	0.11	0.00	0.00	0.15	0.00
01-17-17	0.28	7.11	13.83	0.95	0.00	0.00	1.26	0.00
01-20-17	0.18	4.57	2.98	0.60	0.00	0.00	0.79	0.00
01-21-17	0.10	2.54	1.42	0.32	0.00	0.00	0.42	0.00
01-22-17	1.38	35.05	34.62	4.82	3.27	0.00	6.37	0.00
02-06-17	0.05	1.27	1.45	0.15	0.00	0.00	0.19	0.00
02-07-17	0.31	7.87	2.00	1.06	0.00	0.00	1.40	0.00
02-08-17	0.43	10.92	17.87	1.48	0.00	0.00	1.96	0.00
02-15-17	0.42	10.67	3.58	1.44	0.00	0.00	1.91	0.00

Table D1 Continued

DATE	PRECIPITATION			SOUTH INFLOW	SOUTH OUTFLOW	SOUTH OVERFLOW	NORTH INFLOW	NORTH OVERFLOW
	Depth (in)	Depth (mm)	Duration (hrs)	(m^3)	(m^3)	(m^3)	(m^3)	(m^3)
02-25-17	0.23	5.84	2.17	0.78	0.00	0.00	1.03	0.00
02-28-17	0.56	14.22	4.40	1.94	0.00	0.00	2.56	0.00
03-01-17	1.00	25.40	10.05	3.48	1.43	0.00	4.60	0.00
03-07-17	0.42	10.67	9.45	1.44	0.00	0.00	1.91	0.00
03-10-17	0.56	14.22	5.45	1.94	0.00	0.00	2.56	0.00
03-13-17	0.57	14.48	6.88	1.97	0.00	0.00	2.61	0.00
03-17-17	0.66	16.76	12.17	2.29	0.00	0.00	3.02	0.00
04-03-17	1.62	41.15	18.92	5.66	2.32	0.00	7.49	0.00
04-05-17	0.48	12.20	1.82	1.66	0.00	0.00	2.19	0.00
04-06-17	0.06	1.52	9.77	0.18	0.00	0.00	0.24	0.00
04-12-17	0.05	1.27	0.47	0.15	0.00	0.00	0.19	0.00
04-17-17	0.06	1.52	2.37	0.18	0.00	0.00	0.24	0.00
04-18-17	0.22	5.59	17.50	0.74	0.00	0.00	0.98	0.00
04-19-17	0.04	1.02	2.27	0.11	0.00	0.00	0.15	0.00
04-20-17	0.11	2.79	0.92	0.35	0.00	0.00	0.47	0.00
04-21-17	0.91	23.20	5.23	3.18	0.00	0.00	4.20	0.00
04-22-17	1.18	29.90	10.82	4.11	0.35	0.00	5.43	0.00
04-23-17	1.56	39.60	14.65	5.45	1.21	0.00	7.20	0.00
04-27-17	0.40	10.20	6.32	1.38	0.00	0.00	1.82	0.00
05-01-17	0.19	4.90	2.83	0.65	0.00	0.00	0.85	0.00
05-04-17	0.71	18.10	6.65	2.47	0.00	0.00	3.27	0.00
05-05-17	0.24	6.20	7.60	0.83	0.00	0.00	1.09	0.00
05-06-17	0.26	6.50	5.78	0.87	0.00	0.00	1.15	0.00
06-30-17	0.24	6.10	2.17	0.81	0.00	0.00	1.07	0.00
07-01-17	0.76	19.40	6.98	2.65	0.00	0.00	3.51	0.00
07-04-17	0.28	7.20	5.70	0.96	0.00	0.00	1.27	0.00
07-05-17	0.08	2.10	1.17	0.26	0.00	0.00	0.34	0.00
07-06-17	0.20	5.00	0.27	0.66	0.00	0.00	0.87	0.00
07-23-17	0.22	5.59	1.88	0.74	0.00	0.00	0.98	0.00
07-28-17	1.93	49.10	8.13	6.76	1.60	0.01	8.94	0.00
08-04-17	0.56	14.22	8.85	1.94	0.00	0.00	2.56	0.00
08-06-17	2.87	72.90	24.95	10.06	3.96	0.78	13.30	0.00
08-09-17	0.11	2.79	0.28	0.35	0.00	0.00	0.47	0.00
08-10-17	0.16	4.06	0.63	0.53	0.00	0.00	0.70	0.00
08-11-17	0.09	2.29	2.00	0.28	0.00	0.00	0.38	0.00
08-12-17 AM	0.76	19.30	0.60	2.64	0.36	0.02	3.49	0.00

Table D1 Continued

DATE	PRECIPITATION			SOUTH INFLOW	SOUTH OUTFLOW	SOUTH OVERFLOW	NORTH INFLOW	NORTH OVERFLOW
	Depth (in)	Depth (mm)	Duration (hrs)	(m^3)	(m^3)	(m^3)	(m^3)	(m^3)
08-12-17 PM	0.53	13.46	0.50	1.83	0.10	0.05	2.42	0.00
08-18-17	0.17	4.32	0.52	0.57	0.00	0.00	0.75	0.00
08-27-17	0.22	5.59	0.65	0.74	0.00	0.00	0.98	0.00
08-30-17	0.20	5.08	6.23	0.67	0.00	0.00	0.89	0.00
08-31-17	0.24	6.10	13.78	0.81	0.00	0.00	1.07	0.00
09-01-17AM	0.25	6.35	7.00	0.85	0.00	0.00	1.12	0.00
09-01-17 PM	0.08	2.03	6.22	0.25	0.00	0.00	0.33	0.00
09-02-17	0.14	3.56	7.92	0.46	0.00	0.00	0.61	0.00
09-05-17	1.73	43.90	9.35	6.04	0.38	0.00	7.99	0.00
09-11-17	0.21	5.33	6.27	0.71	0.00	0.00	0.93	0.00
10-08-17	1.91	48.51	21.05	6.68	0.39	0.00	8.84	0.00
10-15-17	0.33	8.38	3.13	1.13	0.00	0.00	1.49	0.00
10-23-17	1.58	40.13	9.33	5.52	1.84	0.01	7.30	0.00
10-28-17	1.01	25.65	13.78	3.52	0.00	0.00	4.65	0.00
11-03-17 AM	0.15	3.81	3.07	0.50	0.00	0.00	0.65	0.00
11-03-17 PM	0.26	6.60	0.58	0.88	0.00	0.00	1.17	0.00
11-06-17	0.10	2.54	5.33	0.32	0.00	0.00	0.42	0.00
11-07-17	0.86	21.84	7.63	2.99	0.13	0.00	3.95	0.00
11-18-17	0.41	10.41	1.50	1.41	0.00	0.00	1.86	0.00
12-05-17	0.87	22.10	13.93	3.03	0.00	0.00	4.00	0.00
12-19-17	1.40	35.56	21.72	4.89	0.51	0.00	6.46	0.00
12-22-17	0.78	19.90	37.02	2.72	0.00	0.00	3.60	0.00
01-11-18	0.09	2.29	4.98	0.28	0.00	0.00	0.38	0.00
01-12-18	0.44	11.18	11.07	1.51	0.00	0.00	2.00	0.00
01-28-18	0.47	11.94	12.37	1.62	0.00	0.00	2.14	0.00
02-01-18	0.33	8.38	8.52	1.13	0.00	0.00	1.49	0.00
02-04-18	0.77	19.50	10.18	2.67	0.00	0.00	3.52	0.00
02-05-18	0.11	2.79	6.55	0.35	0.00	0.00	0.47	0.00
02-07-18	0.74	18.80	4.48	2.57	0.02	0.00	3.40	0.00
02-10-18	2.49	63.25	35.47	8.72	2.06	0.00	11.53	0.00
02-14-18 AM	0.16	4.06	3.73	0.53	0.00	0.00	0.70	0.00
02-14-18 PM	0.20	5.08	7.90	0.67	0.00	0.00	0.89	0.00
02-16-18	0.22	5.59	5.27	0.74	0.00	0.00	0.98	0.00

Table D1 Continued

DATE	PRECIPITATION			SOUTH INFLOW	SOUTH OUTFLOW	SOUTH OVERFLOW	NORTH INFLOW	NORTH OVERFLOW
	Depth (in)	Depth (mm)	Duration (hrs)	(m^3)	(m^3)	(m^3)	(m^3)	(m^3)
02-17-18	0.70	17.78	6.10	2.43	0.06	0.00	3.21	0.00
02-21-18	0.07	1.78	0.57	0.22	0.00	0.00	0.28	0.00
02-24-18	0.10	2.54	5.70	0.32	0.00	0.00	0.42	0.00
02-25-18	0.60	15.24	11.45	2.08	0.00	0.00	2.75	0.00
02-26-18	0.41	10.41	10.85	1.41	0.00	0.00	1.86	0.00
02-28-18 AM	0.17	4.32	5.67	0.57	0.00	0.00	0.75	0.00
02-28-18 PM	1.22	30.99	21.73	4.26	0.02	0.00	5.63	0.00
03-06-18	0.18	4.57	6.07	0.60	0.00	0.00	0.79	0.00
03-11-18 AM	0.13	3.30	5.07	0.42	0.00	0.00	0.56	0.00
03-11-18 PM	0.44	11.18	10.20	1.51	0.00	0.00	2.00	0.00
03-19-18	0.42	10.67	1.85	1.44	0.00	0.00	1.91	0.00
03-24-18	0.79	20.07	20.23	2.74	0.00	0.00	3.63	0.00
03-29-18	0.39	9.91	2.72	1.34	0.00	0.00	1.77	0.00
04-04-18	0.37	9.40	3.93	1.27	0.00	0.00	1.68	0.00
04-15-18	0.89	22.61	24.27	3.10	0.00	0.00	4.09	0.00
04-22-18	1.44	36.58	46.23	5.03	0.00	0.00	6.65	0.00
04-25-18	0.07	1.78	1.05	0.22	0.00	0.00	0.28	0.00
04-26-18	0.13	3.30	9.57	0.42	0.00	0.00	0.56	0.00
05-05-18	0.12	3.05	1.20	0.39	0.00	0.00	0.52	0.00
05-07-18	0.06	1.50	0.72	0.18	0.00	0.00	0.23	0.00
05-15-18	0.06	1.52	1.80	0.18	0.00	0.00	0.24	0.00
05-18-18	0.15	3.81	5.13	0.50	0.00	0.00	0.65	0.01
05-19-18	0.06	1.52	7.08	0.18	0.00	0.00	0.24	0.00
05-22-18	0.25	6.35	4.02	0.85	0.00	0.00	1.12	0.00
05-26-18	0.36	9.14	11.23	1.23	0.00	0.00	1.63	0.00
05-27-18	0.07	1.78	1.55	0.22	0.00	0.00	0.28	0.00
05-28-18	0.20	5.08	6.72	0.67	0.00	0.00	0.89	0.00
05-29-18	0.51	12.95	32.12	1.76	0.00	0.00	2.33	0.00
05-31-18	0.12	3.05	4.15	0.39	0.00	0.00	0.52	0.00
06-03-18	0.87	22.10	0.98	3.03	0.01	0.07	4.00	0.00
06-18-18	0.06	1.52	0.37	0.18	0.00	0.00	0.24	0.00
06-21-18	0.55	13.97	1.95	1.90	0.00	0.00	2.51	0.00
06-26-18	0.76	19.30	22.45	2.64	0.00	0.00	3.49	0.00
06-28-18	1.17	29.72	7.30	4.08	0.33	0.00	5.39	0.00
07-01-18	0.27	6.86	3.22	0.92	0.00	0.00	1.21	0.00

Table D1 Continued

DATE	PRECIPITATION			SOUTH INFLOW	SOUTH OUTFLOW	SOUTH OVERFLOW	NORTH INFLOW	NORTH OVERFLOW
	Depth (in)	Depth (mm)	Duration (hrs)	(m^3)	(m^3)	(m^3)	(m^3)	(m^3)
07-06-18	2.61	66.29	5.18	9.14	5.39	0.31	-	-
07-07-18	0.11	2.79	2.10	0.35	0.00	0.00	0.47	0.00
07-17-18	1.81	45.97	0.97	6.33	1.07	0.29	-	-
07-20-18	1.24	31.50	7.45	4.33	0.31	0.00	5.72	0.00
07-21-18	0.62	15.75	8.00	2.15	0.00	0.00	2.84	0.01
07-22-18	0.09	2.29	8.28	0.28	0.00	0.00	0.38	0.00
07-23-18	1.11	28.19	1.48	3.87	0.82	0.02	5.12	0.02
07-31-18	0.27	6.86	11.32	0.92	0.00	0.01	1.21	0.00
08-01-18	0.57	14.48	16.43	1.97	0.00	0.00	2.61	0.00
08-02-18	0.44	11.18	6.88	1.51	0.00	0.00	2.00	0.00

VITA

R. Andrew Tirpak was born on August 20, 1990 in Youngstown, Ohio to Rick and Nancy Tirpak. He and his younger sister, Meredith, grew up in Columbiana, Ohio, where they lived throughout his childhood. After high school, Andrew attended The Ohio State University in Columbus, Ohio, where he met his future fiancée, Jessie, and graduated with his bachelor's degree in environmental engineering in 2013 with honors. Andrew then went on to work for ExxonMobil in Houston, Texas, where he worked as an environmental, regulatory, and socioeconomics advisor in the Safety, Security, Health and Environment function in the Development Company. In the spring of 2015, Andrew began pursuing his Ph.D. under the direction of Dr. Jon Hathaway at the University of Tennessee, Knoxville in the Department of Civil and Environmental Engineering, from which he graduated in the fall of 2018. Following graduation, Andrew plans to continue his work on green infrastructure with the department as a post-doctoral researcher.